ASSESSING BIOLOGICAL INTEGRITY
IN RUNNING WATER

BOOK

Assessing Biological Integrity in Running Waters
A Method and Its Rationale

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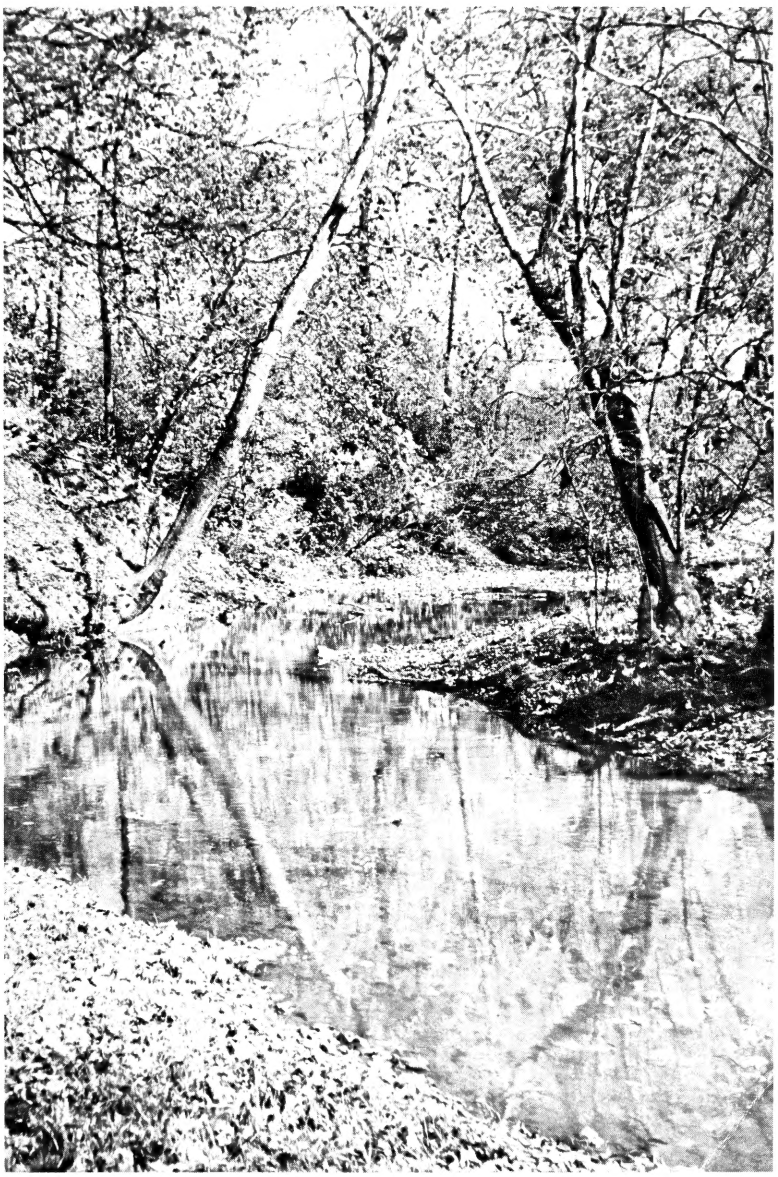

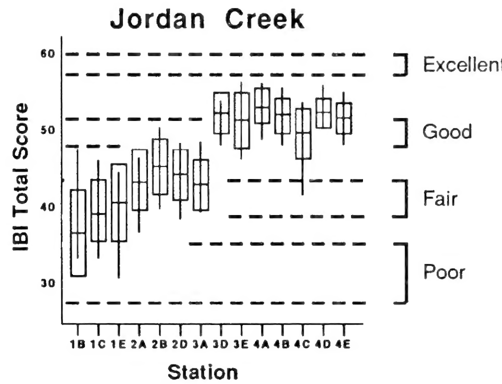

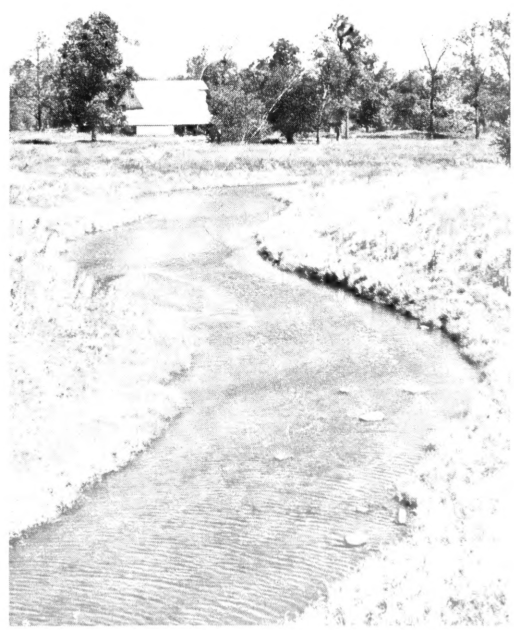

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The Illinois Natural History Survey is pleased to publish this report and make it available to a wide variety of potential users. The Survey endorses the concepts from which the Index of Biotic Integrity was developed but cautions, as the authors are careful to indicate, that details must be tailored to fit the geographic region in which the Index is to be used.
Glen C. Sanderson, Chair, Publications Committee, Illinois Natural History Survey

R. Weldon Larimore of the Illinois Natural History Survey took the cover photos, which show two reaches of Jordan Creek in east-central Illinois—an undisturbed site and a site that shows the effects of grazing and agricultural activity.

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Introduction

The Problem

Concern over the deteriorating quality of water resources in the past several decades has resulted in intervention through federal legislation: the Water Pollution Control Act of 1966, the Federal Water Pollution Control Act Amendments of 1972 (PL 92-500), and the Clean Water Act of 1977 (PL 95-217). A specific charge of that legislation was to restore and maintain the biological, or biotic, integrity of the nation's waters. Perhaps because biotic integrity is difficult to define, efforts to restore the integrity of water resources have been dominated by such nonbiological measures as chemical and physical water quality on the presumption that improvements in biological quality would follow. This approach provides a certain statistical validity and legal defensibility but does not directly measure biological or ecological condition (Herricks and Schaeffer 1985), and we should not be surprised that the biotic integrity of water resources has continued to decline (Gosz 1980; Karr and Dudley 1981; Judy et al. 1984; Karr et al. 1985b).

Recent policy changes by the U.S. Environmental Protection Agency (USEPA) and various state water management agencies acknowledge a natural physicochemical variability in and among bodies of water. Swamps or low-pH bogs, for example, regularly exhibit low dissolved-oxygen levels unrelated to human activity. Recognizing that certain kinds of variability are natural has encouraged us to look more closely at human-induced alterations; however, the focus has remained on chemical monitoring, largely the effluent quality of point sources. Regulations continue to be governed by point-source permits and concentrations of toxic chemicals, and many perturbations that degrade water resources, whether they are caused by humans or occur naturally, are overlooked. Flow alterations and the degradation of physical habitat structure are two examples of human-induced perturbations to which chemical monitoring is insensitive. As a result of our reliance on chemical monitoring, we have often failed to consider the ability of water resources to sustain desirable biological processes at appropriate levels.

A recent nationwide USEPA study found that 56% of the stream segments with water resource degradation had a reduced fishery potential because of chemical problems; however, 50% were impaired by degradation in physical habitat and 67% by flow alteration (Judy et al. 1984). In short, halting the chemical degradation of water does not of itself assure the restoration of its ecological or biotic integrity. The ability of a water resource system to sustain a balanced biological community is obviously the best indicator of its potential; yet that ability is largely unprotected by present monitoring and assessment techniques.

The identification and treatment of the chemical degradation detected by most existing monitoring programs has been dominated by engineering technology. Unfortunately, the lack of appropriate tools for the direct biological assessment of water resources has minimized the participation of aquatic biologists. Even now, as these tools are being developed, they come under attack because they do not work equally well in all situations. They are criticized as "too expensive," "too time consuming," and "subject to gear selectivity." Nevertheless, biological monitoring offers an opportunity to improve and preserve water resources that cannot be ignored, and better tools will be developed, especially if we implement existing programs of biological monitoring in ways to encourage that development. Indeed, a major purpose of this document is to argue that ecologists, aquatic biologists, and ichthyologists must assume major roles in monitoring, evaluating, and managing our water resources. This paper, therefore, demonstrates the need for a method to assess biotic integrity directly, provides a conceptual framework for biological monitoring, and describes a useful tool for biological monitoring—the Index of Biotic Integrity.

Toward a Solution

The Index of Biotic Integrity (IBI) was designed to include a range of attributes of fish assemblages. Its twelve measures, or metrics, fall into three broad categories: Species Composition, Trophic Composition, and Fish Abundance and Condition (Karr 1981). Data are obtained for each of these metrics at a given site and evaluated in light of what might be expected at an unimpacted or relatively unimpacted site located in a similar geographical region and on a stream of comparable size. A number rating is then assigned to each metric based on whether its evaluation deviates strongly from, deviates somewhat from, or approximates expectations. The sum of the twelve ratings, in turn, yields an overall site score. The strength of IBI is its ability to integrate information from individual, population, community, zoogeographic, and ecosystem levels into a single ecologically based index of the quality of a water resource.

A useful feature of IBI is that the collection of data proceeds in progressive steps to a synthetic summary; at the same time, however, IBI preserves for evaluation the data associated with specific biological attributes. This feature enables us to use fully the data obtained during the labor-intensive and cost-intensive collection stage; it also allows us to identify quickly those aspects of community response that may be responsible for a given unsatisfactory rating. Depending on the purpose of the investigation, this identification can lead to further study or to actions that will control or eliminate undesirable conditions. In addition, IBI can be used to screen a large number of sample areas and to determine trends, thus enabling us to assess the success of management programs for water resources. Finally, IBI is based on direct observation, for which there is no reliable surrogate.

A number of researchers and agency personnel have used IBI since its publication (Karr 1981). Some of these uses have been appropriate and others have not. In several, IBI, which was developed for use in the Midwest, was adapted to meet conditions in other regions. We know of uses in two dozen states and provinces, including New York, Virginia, West Virginia, Ohio, Indiana, Illinois, Wisconsin, Tennessee, Kentucky, North Carolina, Minnesota, Wisconsin, Michigan, North Dakota, South Dakota, Nebraska, Arkansas, Missouri, Louisiana, Colorado, Oregon, California, and Ontario. Among its users are federal agencies (Army Corps of Engineers, National Park Service, Soil Conservation Service, U.S. Environmental Protection Agency), state agencies (Illinois, Ohio, Minnesota, Wisconsin), and universities. IBI has been used to gather information in general surveys as well as to examine the impact of specific human actions on water resources, for example, the effects of mine drainage (Leonard 1984; Leonard and Orth 1986), the impact of effluent from sewage treatment plants (Karr et al. 1985a), and a survey of national scenic rivers (Fausch, unpublished data). In the aggregate, these applications demonstrate the ability of IBI to identify a variety of forms of degradation.

Background

The use of the term biotic integrity in water resource legislation (PL 92-500, PL 95-217) is, at best, abstract and somewhat elusive. When tied to ecological systems, the term has been defined as the ability to support and maintain "a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of natural habitat of the region"(Karr and Dudley 1981). Systems possessing biotic integrity can withstand or rapidly recover from most perturbations imposed by natural environmental processes and survive many major disruptions induced by humans. Aquatic systems that lack integrity are often already degraded and when further perturbed by natural and human-induced events are likely to change rapidly to even more undesirable states (Toth et al. 1982). In brief, biotic integrity is possessed by aquatic ecosystems in which composition, structure, and function have not been adversely impaired by human activities. Taken together, chemical, physical, and biotic integrity can be equated with ecological integrity.

Biotic integrity, however, is not necessarily correlated with harvestable products or services of economic value. Indeed, the presence in some systems of harvestable products in large amounts may indicate a loss in integrity.

Although the importance of biotic integrity is both implicit and explicit in water resource legislation, the biological assessment of water resources has been hampered by a lack of theory supported by empirical data on which to base a method for evaluating biological conditions in a broadly based and integrative context, by the uncritical acceptance of chemical monitoring to assess biological status, and by the tendency of wildlife agencies to focus on single-species management.

Factors That Affect Biotic Integrity

The biotas of streams have evolved over millions of years. Although many environmental factors have been instrumental in the evolution of those biotas, recent studies suggest that these factors can be grouped into five major classes (Fig. 1). Altering the physical or biological processes associated with any of these classes typically has a major impact on stream biota and, thus, on biotic integrity. Efforts to restore or maintain water resource quality by altering factors in only one of these classes—for example, water quality—will fail if factors in another class—for example, habitat structure—limit biotic integrity (Gorman and Karr 1978; Karr and Dudley 1978; Karr and Schlosser 1978; Karr and Dudley 1981). Those who would improve the biological integrity of a waterway, therefore, must have the means to identify perturbed processes associated with factors in all of these classes.

Although the landscape pattern of a watershed—its regional topography, soil and vegetation types, and land use—determines in large measure the nature of these factors, human activity alters this temporal and spatial landscape pattern and thus profoundly affects the biotas of streams and rivers. The effects of human activity vary among streams depending in part on their size; however, as Figure 1 makes clear, general patterns of degradation follow man's perturbations. Figure 1 also indicates why single-factor or single-class approaches generally fail to achieve long-term biotic integrity. Not only are broad-based approaches more likely to solve water resource problems but they are also more likely to prove cost effective because they often capitalize on natural cleaning processes in much the same way that secondary treatment purifies sewage effluent. Thus, an appropriate solution to a given set of water quality problems may be to develop a riparian and instream habitat that is structurally suited to a rich biota. The effects of soluble nutrients, which often produce blooms of nuisance algae in stream channels, can, for example, be reduced by ensuring that streams have overhanging cover. Shading restricts light and thus limits the growth of algae. Lowered production of algae, in turn, affects the aquatic invertebrate community and the processing of organic matter.

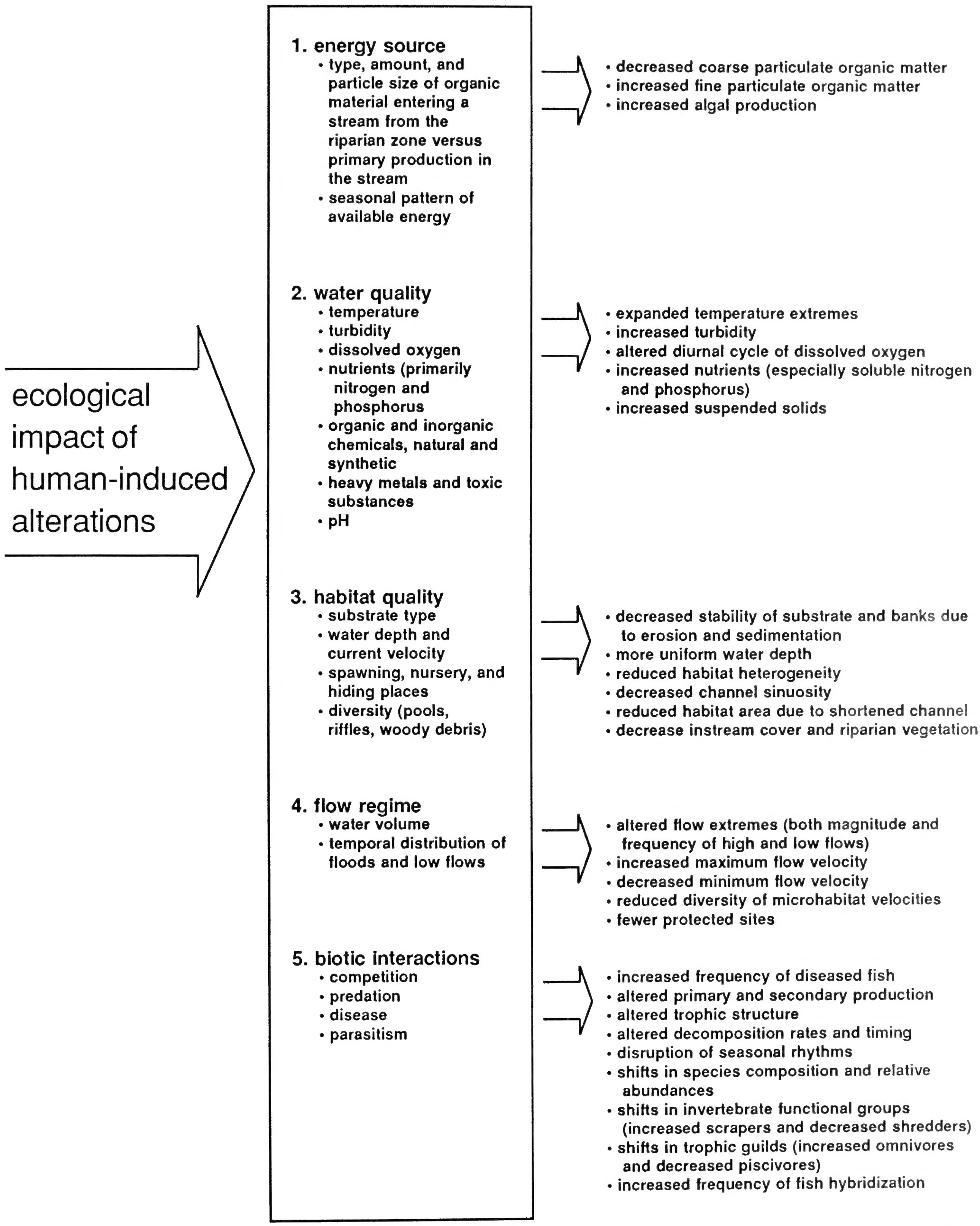

Fig. 1. Five major classes of environmental factors that affect aquatic biota. Arrows indicate the kinds of effects that can be expected from human activities, in this case the alteration of headwater streams, excluding small impoundments (modified from Karr et al. 1983).

Approaches to the Evaluation of Biotic Integrity

Biologists have taken several approaches to biological assessment. Perhaps the most common relies on indicator species to identify low or high quality. One such approach is Hilsenhoff's tolerance index (1977, 1982) using benthic invertebrates. The applicability of this index is limited, however, because invertebrates are sensitive to relatively few types of degradation, for example, depleted oxygen. Whether or not benthic invertebrates can be used to identify the degradation of physical habitat or the presence of toxicants is not well established. In addition, the tolerances of aquatic invertebrates have not been precisely defined in many areas, especially for insect taxa in which species are difficult to distinguish.

Ryder and Edwards (1985) recently advocated the use of indicator organisms and defined a number of characteristics that such organisms must possess. Despite the different tolerances and thresholds that occur among the numerous stocks or races of lake trout, *Salvelinus namaycush*, they selected that species as a primary indicator.

Schaeffer et al. (1985) suggested that indicator organisms be used in screening programs for subsequent, more definitive chemical studies. They err, however, in assuming that the only cause of water resource degradation is chemical imbalance and that chemical metrics are superior to biological metrics in the assessment of biological potential. Although indicator species are useful in selected cases, the widespread applicability of this approach has not been demonstrated and, at least in the case of Schaeffer et al., its theoretical bias is troubling.

Although their work involved bird rather than aquatic populations, Verner (1984) and Morrison (1986) are among the researchers who have experienced difficulties in the use of indicator species.

An approach to biological assessment that gained popularity in the 1960s is based on the computation of one or more of a variety of diversity indexes. The rationale for its use is that environmental perturbation leads to a reduction in the number of species and to dominance by a small number of tolerant species (Patrick 1949). Diversity indexes combine two attributes of a collection: number of species and evenness, or the degree to which all species present are equally represented (Wilhm and Dorris 1968; Pielou 1975). Although many researchers have used numerical abundance for the second component, biomass has also been used in diversity calculations, and some workers have proposed an hierarchical approach to the calculation of diversity indexes at ordinal, familial, generic, and species levels (Kaesler et al. 1978; Osborne et al. 1980).

Gammon et al. (1981) used diversity indexes to evaluate fish communities in the Wabash River, combining information about number of individuals and biomass per km of stream with Shannon-Weiner diversity indexes based on number of individuals and biomass per species. The index of community well-being that resulted has also proved valuable in tests in the Willamette River of Oregon (Hughes and Gammon, unpublished manuscript). Unfortunately, biomass data are twelve times more costly to collect than species enumeration data and six times more costly than abundance data. Further, the field procedures used to weigh the wide range of fish encountered in stream studies are often inaccurate (Johnson and Nielsen 1983:15). (Mass weighing of small species decreases the cost of biomass estimates by an unknown amount.)

Although the indexes described above have been empirically useful (Hendricks et al. 1980), combining species richness with species abundance or biomass in a single index can yield ambiguous results that may prove difficult to interpret (Kovalak 1981). In streams with low species richness, as are common in the West, an increase in richness often means that more exotic species or species that are more tolerant and less desirable have been added. Degradations that change a community with many species, some of which exist in relatively low abundance, to a community with a few species that are relatively equal in abundance may actually increase the diversity index, even when the total number of individuals has been reduced. In the absence of other information, such increases in diversity might easily be misconstrued as improvement in conditions.

A third approach to biological assessment is based on the relative abundance of desirable species, for example, the percentage of sport fishes (Coble 1982). This strategy, unlike that used with indicator species, requires long-term data bases to ensure that the species chosen are valid indicators of environmental conditions. Limiting analysis to desirable species has three drawbacks. First, guidelines based on long-term research are necessary, and they do not exist for most streams, especially small warm-water streams. Second, the definition of "desirable" varies among cultures and even within regions of the United States. Coble (1982), for example, excluded suckers (Catostomidae), bullheads (Ictaluridae), and common carp from his list of sport fishes. Swink and Jacobs (1983), however, included some suckers on their list, and bullheads and common carp are considered sport fishes elsewhere. Third, valuable information about the rest of the community may be ignored when desirable species are used as the assessment criterion, thereby decreasing the efficiency with which the limited available data are used. Analyses based on desirable species, although not useful in the assessment of biotic integrity, may be helpful at sites governed by narrowly defined fishery management plans.

More recently, the U.S. Fish and Wildlife Service has measured physical habitat conditions using the Habitat Evaluation Procedure (Terrell et al. 1982) and the Instream Flow Incremental Methodology (Stalnaker 1982). A fundamental assumption in choosing these models is that fish populations are limited by the availability of suitable habitat. Accumulating background information for use with these models is costly, however, and researchers tend to estimate or to assume critical parameter values. Further, little or no provision is made for geographic variability in the habitat requirements of a given species. Finally, like water quality data acquired through chemical monitoring, habitat data, which are based on expected population performance under "normal" habitat conditions, can, at best, be used only indi-

rectly to assess biotic integrity. In our view, the direct assessment of broader attributes of the biological community is to be preferred. The habitat measurements used by the U.S. Fish and Wildlife Service, however, are helpful where management goals focus on selected target species, for example, salmonids, rather than on biotic integrity.

Fish as Biological Monitors

Biological communities reflect a combination of current and past watershed conditions because organisms are sensitive to changes across a wide array of environmental factors. Many groups of organisms have been proposed as indicators of environmental quality, but no single group has emerged as the favorite of most biologists. Indeed, under ideal circumstances, a biological monitoring program might be based on a holistic, integrative approach that incorporates representatives of several major taxa. Limited funds and time, however, usually dictate more restrictive approaches.

Diatoms (Patrick 1975) and benthic invertebrates (Resh and Unzicker 1975; Hilsenhoff 1977; Mason 1978) have frequently been used for biological monitoring; however, their use presents certain difficulties. Because diatoms are difficult and time consuming to identify and sort, technicians may require more specialized taxonomic expertise. In addition, life-history information is lacking for many species and groups. Diatoms are greatly affected by microhabitat conditions on the scale of a meter or so, and their frequent reproduction and resulting recruitment can also mask temporal impacts. Further, diatoms can be used to assess stream conditions only for energy producers and not for energy consumers. Finally, the information obtained by using diatoms and invertebrates as biological monitors is often difficult to convey to the general public. Although useful water resource decisions have been made based on the use of diatoms and insects, we maintain that more informed and less costly decisions are possible when fish are used as the primary taxon in biological monitoring. Their advantages as indicator organisms in the assessment of biotic integrity are summarized below (modified from Karr 1981).

To begin, fish are typically present even in the smallest streams and in all but the most polluted waters. They occupy positions throughout the aquatic food web and thus provide an integrative view of watershed conditions. Fish communities generally include species that represent a variety of trophic levels (omnivores, herbivores, insectivores, planktivores, piscivores), and their diets often include foods from both the terrestrial and aquatic environments. Since many species reproduce only once a year and at an established spawning season, populations are relatively stable during the summer when most sampling activities occur (excluding fry less than 25 millimeters long). Because fish often range considerable distances, they have the potential to integrate diverse aspects of relatively large-scale habitats. Because they are comparatively long-lived, they permit a temporal dimension in the assessment of stream conditions. Careful examination of recruitment and growth data among years can help to pinpoint periods of unusual stress. Both acute toxicity (missing taxa) and stress effects (depressed growth and reproductive success) can be evaluated. In general, fish species are primarily affected by macro-environmental influences; algae and invertebrates are more subject to both micro and macro-environmental influences.

Compared to diatoms and invertebrates, fish are relatively easy to identify, and training workers is a less difficult task. Indeed, most samples can be identified and sorted at the field site and then released. In most cases, no long-term laboratory work is required—work that is often delayed due to other demands. (How many invertebrate samples remain unprocessed on laboratory shelves?)

Fortunately, life-history information is extensive for many fish species, especially commercial and sport fishes; at least some information is available for virtually all North American species. These data, however, are not always adequately archived, and much has not been sufficiently analyzed. Population and community data on fish are widely collected each year by fish and game departments, university ichthyologists, and others interested in stream biology. Typically, these data bases are overlooked or inadequately used in environmental evaluations. At issue is not the availability of life-history information but how we can improve the quality of that data and how best to use it.

Since public law refers to fishable waters, citizens in general are more likely to understand information about the condition of the fish community than data on inveterbrates. In addition, the results of studies in which fish were used as indicator species can often be directly related to the protection of aquatic biotas as mandated by Congress. Finally, fish communities are valuable economic resources that should be monitored and maintained for their own sake.

In spite of these advantages, fish have rarely been used in comprehensive monitoring programs (but see Hocutt and Stauffer 1980). They are, however, commonly used in bioassays (Sprague 1973), typically the bioassay of contaminants, often using representative species (USEPA 1977). The field monitoring of fish has also been directed toward harvests of sport or commercial species.

The use of fish as biological monitors is, of course, not without difficulties. Among these are the selective nature of sampling gear for certain sites and for certain sizes and species of fishes, the mobility of fish on diel and seasonal time scales, and the number of technicians needed for field sampling. Nevertheless, problems of equal or greater magnitude are associated with the use of other taxa. Indeed, problems not unlike those noted above are commonly found in chemical monitoring, for example, differences in water samples taken at various times of the day or differences in samples taken at the edge versus the center of a channel or at the surface as opposed to the subsurface.

A New Method: The Index of Biotic Integrity

The accurate assessment of biotic integrity requires a method that integrates biotic responses through an examination of patterns and processes from individual to ecosys-

tem levels. One tactic is to define an array of biological metrics much like the economic indicators used in econometric analyses. The Index of Biotic Integrity (IBI) adopts this tactic, incorporating data from the study of entire fish communities in twelve metrics in three categories (Table 1). The value of each metric is then compared to the value expected at a site located in a similar geographical region on a stream of similar size where human influences have been minimal (Weber 1981). Ratings of 5, 3, and 1 are assigned to each metric according to whether its value approximates, deviates somewhat from, or deviates strongly from the value expected at the relatively undisturbed site. The sampling site is then assigned to one of six quality classes based on the total of the twelve metric ratings. The highest score, 60, indicates a site without perturbation; sites of reduced quality have lower scores. These scores are also given qualitative labels that range from excellent to very poor (Table 2).

IBI metrics assess attributes that are assumed to correlate with biotic integrity, which is itself an abstract concept that cannot be measured directly. Individually, each metric provides information about a specific attribute of the sampling site. Together, they characterize the underlying biotic integrity of that site. The values of the twelve metrics, however, are functions of the underlying biotic integrity; biotic integrity is not a function of the metrics.

Measuring the biotic integrity of a stream is in a sense analogous to measuring human health. When blood pressure readings, white blood cell counts, and the results of stress tests fall within acceptable ranges, good health is indicated. Good health, however, is not a simple function of these attributes. Rather, a biological system—whether it is a human system or a stream ecosystem—can be considered healthy when its inherent potential is realized, its condition is stable, its capacity for self-repair when perturbed is preserved, and minimal external support for management is needed.

In summary, IBI relies on multiparameters, a requirement when the system to be evaluated is complex. It incorporates professional judgment in a systematic and sound manner, but it also sets quantitative criteria that enable us to determine what is excellent and what is poor. Admittedly, some criteria are more difficult to implement than others; for example, metric 10, the number of individuals in the sample, is most reliable when a relatively high catch per unit of effort occurs. Similarly, the expectation criteria used to rate the metric data vary with stream size and region (Fausch et al. 1984) and must be adjusted for given fish faunas.

IBI, like any tool, must be used appropriately. It is designed to be used only when the objective is to monitor biotic integrity at specific sites. It is suited for screening a large number of sites in order to identify those that require attention and for assessing trends over time at an individual site. When other objectives are pursued, for example, the management of a single species, the index is of little value. IBI is most appropriately used to interpret large amounts of data from complex fish communities when the objective is to assess biotic integrity.

An especially useful feature of IBI is that it enables researchers to formalize the professional judgments they

Table 1. Metrics used to assess fish communities in the midwestern United States (modified from Karr 1981 and Fausch et al. 1984).

Category	Metric	Scoring criteria[a] 5	3	1
Species richness and composition	1. Total number of fish species	Expectations for metrics 1-5 vary with stream size and region and are discussed in the text.		
	2. Number and identity of darter species			
	3. Number and identity of sunfish species			
	4. Number and identity of sucker species			
	5. Number and identity of intolerant species			
	6. Proportion of individuals as green sunfish	<5%	5-20%	>20%
Trophic composition	7. Proportion of individuals as omnivores[b]	<20%	20-45%	>45%
	8. Proportion of individuals as insectivorous cyprinids	>45%	45-20%	<20%
	9. Proportion of individuals as piscivores (top carnivores)	>5%	5-1%	<1%
Fish abundance and condition	10. Number of individuals in sample	Expectations for metric 10 vary with stream size and other factors and are discussed in the text.		
	11. Proportion of individuals as hybrids	0%	>0-1%	>1%
	12. Proportion of individuals with disease, tumors, fin damage, and skeletal anomalies	0-2%	>2-5%	>5%

[a]Ratings of 5, 3, and 1 are assigned to each metric according to whether its value approximates, deviates somewhat from, or deviates strongly from the value expected at a comparable site that is relatively undisturbed.

[b]Omnivores are defined here as species with diets composed of ≥25% plant material and ≥25% animal material.

make in assessing biotic integrity. IBI, however, does not result in assessments that are more subjective than those obtained by such seemingly objective methods as diversity indexes or criteria for physical water quality. Judgments, of course, are made in establishing the criteria used to rate IBI metrics; however, diversity indexes also rely on judgment when researchers and managers decide how an index value answers questions concerning environmental quality. Unfortunately, the use of seemingly objective indexes sometimes encourages the abrogation of professional judgment and the acceptance of theoretically and empirically tenuous criteria. In fact, all metrics used for assessing the quality of a water resource or for setting water quality standards are subjective, including the establishment of tolerable chemical thresholds (Thurston et al. 1979).

The Metrics

The twelve IBI metrics reflect insights from several perspectives to the study of aquatic biota: individual, population, community, ecosystem, and zoogeographic. Although the metrics are sometimes redundant because several may be sensitive to the same impact, in the aggregate they appear to be responsive to changes of relatively small magnitude as well as to broad ranges of environmental degradation. As Figure 2 indicates, some are sensitive across the range of integrity; others are sensitive to only a portion of that range. If, for example, the number of darter species at a given site declines to zero at an intermediate level of integrity, the abundance metric cannot be used to distinguish differences throughout the range of low quality. Instead, the metric that records the relative abundance of diseased individuals might prove more useful. Our work thus far suggests that the relative sensitivity of a given metric varies from region to region and is relative to the scale of the study; no single metric is always a reliable indicator of degradation (Angermeier and Karr 1986; Karr et al. in press).

Because the metrics are differentially sensitive to various perturbations—siltation or toxic chemicals, for example—as well as to various portions of the range of integrity, conditions at a site can be determined with considerable accuracy. Karr et al. (1984), for instance, found that municipal effluents depressed total numbers of fishes and altered the trophic structure of the community. Habitat modifications, however, most affected a particularly sensitive taxonomic group, the darters.

The remainder of this section is given over to a discussion of the ecological basis of each metric. The twelve metrics are introduced by category and in the order shown in Table 1. This discussion is of particular interest to those who would adapt the metrics to geographical regions outside the Midwest.

Species richness and composition. This category assesses species richness and, to some extent, species composition in comparison to stream size and zoogeographic factors. Expectation values for species richness in undisturbed areas should be based on region, stream size, elevation, and stream gradient. Both total number of species and number of intolerant species are considered along with species in three major families: suckers (Catostomidae), sunfishes (Centrarchidae), and darters (Percidae). Suckers and darters feed predominantly on benthic invertebrates, but sunfishes feed primarily on midwater and surface invertebrates. The sen-

Table 2. Total IBI scores, integrity classes, and the attributes of those classes (modified from Karr 1981).

Total IBI score (sum of the 12 metric ratings)	Integrity class	Attributes
58-60	Excellent	Comparable to the best situations without human disturbance; all regionally expected species for the habitat and stream size, including the most intolerant forms, are present with a full array of age (size) classes; balanced trophic structure.
48-52	Good	Species richness somewhat below expectation, especially due to the loss of the most intolerant forms; some species are present with less than optimal abundances or size distributions; trophic structure shows some signs of stress.
40-44	Fair	Signs of additional deterioration include loss of intolerant forms, fewer species, highly skewed trophic structure (e.g., increasing frequency of omnivores and green sunfish or other tolerant species); older age classes of top predators may be rare.
28-34	Poor	Dominated by omnivores, tolerant forms, and habitat generalists; few top carnivores; growth rates and condition factors commonly depressed; hybrids and diseased fish often present.
12-22	Very poor	Few fish present, mostly introduced or tolerant forms; hybrids common; disease, parasites, fin damage, and other anomalies regular.
	No fish	Repeated sampling finds no fish.

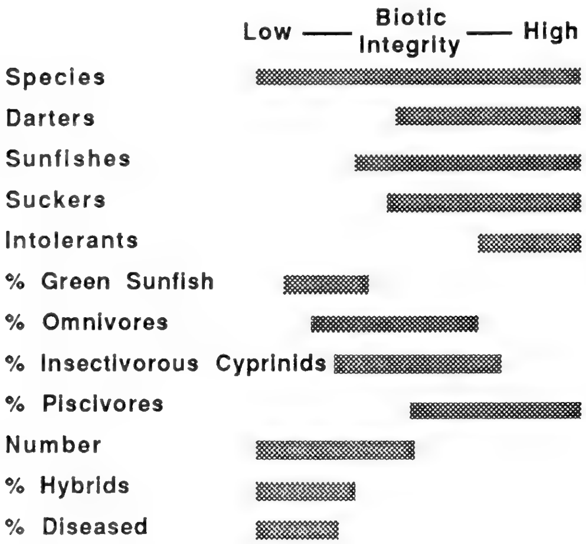

Fig. 2. Range of primary sensitivity for each of the twelve IBI metrics listed in Table 1 (after Angermeier and Karr 1986).

sitivity of these taxa/ecological groups to degradation of their food resources and habitats makes them ideal indicator species for IBI. In addition, their relatively high species richness and broad geographical distributions make them useful monitors of ecosystem degradation over a wide range of conditions. Recent data sets from regions outside the Midwest indicate that other families may need to be substituted, especially when an entire family is missing from a region.

In regions where these taxa are not present, other relatively species-rich groups with wide geographic distribution should be substituted, including one benthic- and one nonbenthic-oriented taxon. The third family can be either, with a minor preference for benthic orientation because the degradation of aquatic systems often disproportionately affects benthos. Sampling should also be taken into consideration when choosing taxa. For samples from larger rivers, appropriate taxa might include groups with larger species—suckers, catfishes, and sunfishes, for example. Guidelines for determining expectation values for species richness and abundance (metrics 1 through 5) are provided in Fausch et al. (1984) and are included in Appendix I.

Metric 1. Total number of fish species. If other features are similar, the number of fish species supported by streams of a given size in a given region decreases with environmental degradation. Hybrids and subspecies are not included in this count. The treatment of exotic species is discussed later.

Metric 2. Number of darter species. The number of species present in the subfamily Etheostomatinae (darters) of the family Percidae provides the data for this metric. These species are sensitive to degradation, particularly as a result of their specificity for reproduction and feeding in benthic habitats (Page 1983). Such habitats are degraded by channelization, siltation, and reduction in oxygen content. For regions outside the range of darters, the number of sculpin species (Cottidae), or another taxon of benthic fishes, may be substituted.

Metric 3. Number of sunfish species. Members of the family Centrarchidae, exclusive of black basses (*Micropterus*), are used because they are particularly responsive to the degradation of pool habitats and to such other aspects of habitat structure as instream cover (Gammon et al. 1981; Angermeier 1983). Other pool-dwelling taxa—for example, salmonids—should be substituted where a diversity of sunfishes does not occur.

Metric 4. Number of sucker species. All members of the family Catostomidae are included because many such species are intolerant of habitat and chemical degradation. In addition, the longevity of suckers provides a multiyear integrative perspective.

Metric 5. Number of intolerant species. Species of many families are intolerant of a variety of perturbations—water quality or habitat degradation or a combination of the two such as loadings of high suspended solids and the siltation that results. Intolerant species are among the first to be decimated after perturbation (see regional ichthyological references such as Pflieger 1975; Smith 1979; Trautman 1981; Becker 1983), and the species identified in metrics 2-4 may be included in this group. Endangered or threatened species, however, are not automatically considered intolerants because their low numbers may be due to factors other than perturbation. They might, for example, be glacial relics. If a high number of intolerant species is included in this metric, its usefulness is reduced because intolerants are most often found only when stream conditions are good to excellent. Overall, the intolerant class should be restricted to the 5 to 10% of species that are most susceptible to such major types of degradation as siltation, lowered flow, low dissolved oxygen, and toxic chemicals. One means of identifying intolerant species is to compare recent species accounts with those collected several decades earlier. If dramatic reductions in abundance or range are apparent, intolerance may be assumed. By convention, species judged intolerant should have disappeared, at least as viable populations, by the time the stream has been degraded to the "fair" category.

Our analyses to date suggest that species sensitive to habitat degradation, especially siltation, are most likely to be identified as intolerant. This phenomenon may well stem from the extensive landscape modifications that have affected midwestern streams. In areas where other types of disturbances are dominant—mine runoff in West Virginia or irrigation withdrawals in the West—other types of species would be considered most intolerant.

Metric 6. Proportion of individuals as green sunfish. In the Midwest, the green sunfish (*Lepomis cyanellus*) increases in relative abundance in degraded streams and may increase from an incidental to the dominant species. It is, therefore, an appropriate species for this metric, which evaluates the degree to which typically tolerant species dominate the community. Other tolerant species—carp, goldfish, and black bullhead—that are often present in moderate numbers but can become dominant in degraded locations can be substituted for green sunfish. Another method of obtaining data for metric 6 is to list tolerant species and compute the pro-

portion of individuals that these species represent. This approach avoids weighting this metric so heavily on a single species.

Trophic composition. The energy base and the trophic dynamics of a stream community are assessed by the three metrics in this category. All organisms require reliable sources of energy, and major efforts have been made to measure the many dimensions of productivity and the trophic structure they produce. These efforts, however, have generally proved costly and time consuming and have often produced ambiguous results. (Is high algal production, for example, desirable or undesirable?) Thus, a means was needed to measure the divergence from expected production and consumption patterns that perturbation causes. The trophic structure of the community can provide this information because alterations in water quality or other habitat conditions, including land use in the watershed, commonly result in changes in the fish community due to fluctuating food resources. The metrics in this category measure these alterations in community function. Species are assigned to the trophic groups based on the feeding patterns of freshwater adults. Three major trophic groups—omnivores, insectivorous cyprinids, and piscivores—are used in metrics 7, 8, and 9.

At present, we lack the information necessary to assign scores to these metrics over a wide range of geographic areas. Outside the Midwest, the values offered here should be applied cautiously until further studies establish their generality.

Metric 7. Proportion of individuals as omnivores. We consider as omnivores species that take significant quantities of both plant and animal materials (including detritus) and have the ability, usually indicated by the presence of a long gut and dark peritoneum, to utilize both. For our purposes here, we follow Karr (1971) for birds and Schlosser (1982) for fish in defining omnivores as species whose diets contain at least 25% plant and 25% animal foods. Precise data are rarely available on the proportions of animal and plant foods for individual species, and variations in time and space are likely. We do not include as omnivores species that may take a variety of animal material but take no plants; neither do we include species with short guts that occasionally contain plants, for example, bluegills (*Lepomis macrochirus*) and creek chubs (*Semotilus atromaculatus*).

The common omnivores of small midwestern streams are *Pimephales notatus* and *P. promelas*; *Cyprinus carpio* and *Dorosoma cepedianum* are found over a wider range of stream sizes. Highly degraded streams also may support large populations of the omnivorous goldfish (*Carassius auratus*). The dominance of omnivores occurs as specific components of the food base become less reliable; the opportunistic foraging habits of omnivores make them more successful than specialized foragers.

Metric 8. Proportion of individuals as insectivorous cyprinids. Metric 8 tends to vary inversely with metric 7. Most North American cyprinids are insectivores (Carlander 1969, 1977). Although insectivorous cyprinids are a dominant trophic group in midwestern streams, their relative abundance decreases with degradation, probably in response to variability in the insect supply, which in turn reflects alterations of water quality, energy sources, or instream habitat. In large rivers and in the southeastern United States and in other regions where insectivorous cyprinids are not as dominant as they are in the Midwest, the proportion of total insectivores to total individuals may provide better information for this metric. This alternative, of course, requires resetting the scoring criteria outlined in Karr (1981) and Fausch et al. (1984). When Angermeier and Karr (unpublished data) used proportion of individuals as insectivores, the classes were 0-40%, >40-80%, and >80% for ratings 1, 3, and 5, respectively.

Metric 9. Proportion of individuals as piscivores. This metric includes individuals of all species in which the adults are predominantly piscivores. Some feed on invertebrates and fish as fry and juveniles. We do not include species like the creek chub that may opportunistically eat some fish, especially as large adults (Fraser and Sise 1980). Viable and healthy populations of such top carnivore species as smallmouth bass (*Micropterus dolomieui*), walleye (*Stizostedion vitreum*), and pike (*Esox* spp.), for example, indicate a healthy, trophically diverse community. Some species in this group may feed extensively on crayfish and frogs.

Fish abundance and condition. The three metrics in this category evaluate such attributes of populations as abundance, age structure, growth and recruitment rates, and fish condition. Because of time and budget constraints, general rather than detailed attributes are measured, as was the case with the productivity metrics described above.

Metric 10. Number of individuals in a sample. This metric evaluates populations and is expressed as catch per unit of sampling effort. Effort may be expressed per unit of area sampled, per length of reach sampled, or per unit of time spent. In streams of a given size and with the same sampling method and efficiency of effort, poorer sites are generally expected to yield fewer individuals than sites of higher quality. Relative catch per unit of effort, therefore, is used to assign scores among sites or at the same site sampled in similar ways at different times. Some disturbances may cause a general decrease in numbers of individuals even though other community characteristics do not change (Kovalak 1981).

Based on the empirical relationship of density as inversely related to watershed area, Miller et al. (unpublished manuscript) propose to establish scoring criteria for this metric with a maximum density line similar to the maximum species richness line of Fausch et al. (1984). This approach has promise but requires testing and evaluation.

Metric 11. Proportion of individuals as hybrids. This metric is difficult to determine from historical data and is sometimes omitted for lack of data. Its primary purpose is to assess the extent to which degradation has altered reproductive isolation among species.

Environmental degradation can lead to increased frequency of hybridization, probably as a result of habitat degradation that reduces segregation of breeding fish along normal habitat gradients such as substrate types (Hubbs

1961). Hybridization may be common among cyprinids after channelization (Greenfield et al. 1973), although difficulties in recognizing hybrids may preclude using this criterion with darters as well as cyprinids. Sunfishes also hybridize quite readily, and the frequency of their hybridization appears to increase with stream modifications.

Metric 12. Proportion of individuals with disease, tumors, fin damage, and skeletal anomalies. Sites with especially severe degradation often yield a high number of individuals in poor health (Mills et al. 1966; Brown et al. 1973; Baumann et al. 1982). Parasitism has been shown to reflect both poor environmental condition and reduction in reproductive capacity (sterility) in fish (Mahon 1976). Indications of poor health include tumors, fin damage or other deformities, heavy infestations of parasites, discoloration, excessive mucus, "redness," and hemorrhaging. To date, few reliable data exist for setting criteria for this metric. Even in pristine areas, a small incidence of anomalies is to be expected; however, sites where these problems are more common are generally located in more degraded areas.

Rating the Metrics and Classifying the Site

Collecting and interpreting IBI information is an hierarchical process (Fig. 3). It begins with the definition of the fish community to be studied and the choice of an appropriate sampling design. Before fish are sampled and their numbers recorded, all species in the regional fish fauna must be characterized according to food requirements and tolerance status. In this section we describe the scoring and classification activities that follow tabulation. Sampling methods are discussed in the section that follows.

After data from the sampling sites have been collected, values for the twelve metrics are compared with their corresponding expectation values and a rating of 5, 3, or 1 is assigned to each metric (Table 1). The sum of these ratings, the total IBI score, is then used to provide a qualitative label for the site (Table 2).

The expectation criteria used to rate each metric must reflect the stream size and the geographical region of the sampling site. Although some of these criteria vary only slightly among stream communities, values for metrics 1-5 in the first category—Species Richness and Composition—vary substantially with stream size and region. The data collected for each of these metrics, therefore, must be compared with data representative of "excellent" fish communities in unperturbed sites on a stream of similar size in the same geographic region.

Because the total number of fish species and the number of species in the three key taxa (metrics 1-4) increase with stream size, the definition of stream size is of considerable importance in establishing expectation criteria. A classification system developed by Horton (1945) and modified by Strahler (1957) is commonly used by aquatic biologists to indicate progressive increases in stream size. According to this system, the smallest streams in a watershed are first order. When two streams of the first order join, they form a stream of the second order; when two second-order streams join, they form a third-order stream. Ecological discussions of streams typically rely on three classes of size: headwaters, streams of the first, second, and third orders; intermediate-sized rivers, streams of the fourth through the sixth order; and large rivers, those of the seventh and larger orders (Vannote et al. 1980). Although this classification is generally useful, the effects of stream order vary among watersheds. Differences in climate, geology, and watershed geomorphology, for example, affect the nature of the stream-order pattern (Hughes and Omernik 1981, 1983) in certain situations, and thus the area of the watershed may be a more useful definition of size than stream order.

When the total number of fish species as a function of stream order or watershed area for a number of relatively undisturbed sites within a watershed are plotted, the points produce a distinct right triangle, the hypotenuse of which approximates the upper limit of species richness (Fausch et al. 1984 and Fig. 4). Sites with migratory species and fish that have escaped from reservoirs are excluded from the count. We judge that a line with slope fit by eye that includes about 95% of the sites is a better measure of expected species

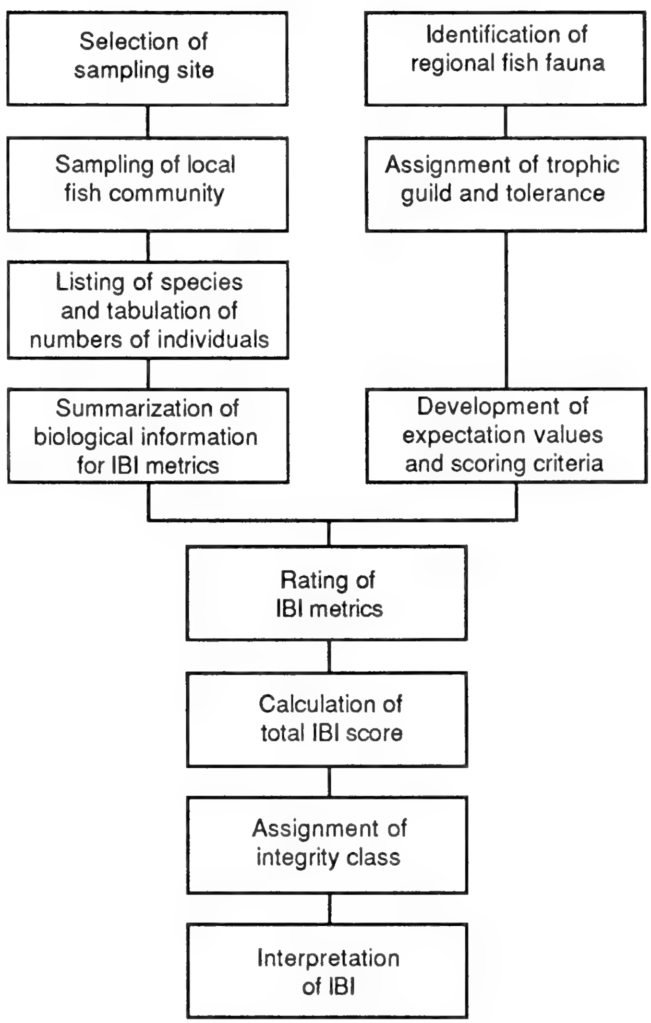

Fig. 3. Sequence of activities involved in calculating and interpreting the Index of Biotic Integrity for a stream segment.

richness over stream size than one provided by a linear regression. We use this line, the line of maximum species richness, to define an "excellent" fish community for purposes of rating metrics 1-5 in the first IBI category (Table 1). Thus when metric 1—total number of species—is rated, the line of maximum species richness is used as a criterion to determine whether species richness for a given site on a stream of a given order or watershed area approximates, is somewhat less than, or is far less than the species richness expected for an "excellent" fish community in that region.

Similar lines are drawn for the three major taxa (darters, sunfishes, and suckers, Table 1) and for intolerant species, but a general knowledge about the fish communities of a particular region must also be taken into account. If, for example, no sunfishes inhabit small streams in a region, a rating of 5 is arbitrarily assigned to metric 3 for small stream sites because the absence of sunfishes does not indicate a degraded condition. Lines of maximum richness for specific taxa are unlikely to be smooth, especially when stream order is used to plot the points, because fewer species are involved (Fausch et al. 1984). Further, the data used to plot these lines must be based on individual samples. Because stream fish communities are dynamic, combining several samples taken from one site on more than one date can lead to erroneous conclusions and false expectations about species richness.

Metrics in the remaining two IBI categories—Trophic Composition and Fish Abundance and Condition—appear to vary less with watershed area, stream size, and geographic region, and we have not yet determined whether that variation is both systematic and large enough to warrant adjusting the expectation criteria. A more careful examination of this variation should be undertaken, especially in studies outside the midwestern United States.

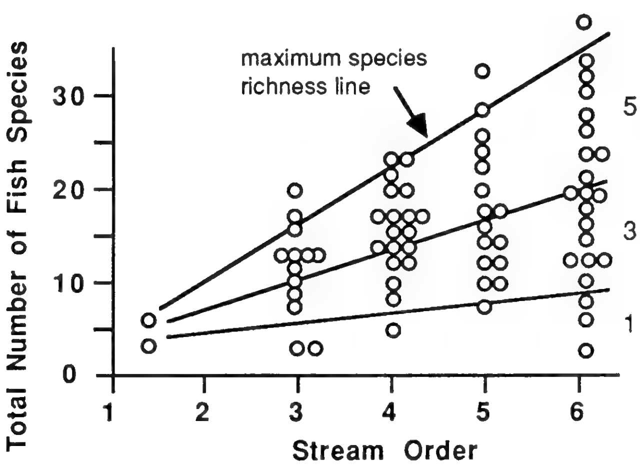

Fig. 4. Total number of fish species versus stream order for 72 "least disturbed" sites along the Embarras River in Illinois. The area below the line of maximum species richness is trisected and used to rate IBI metric 1, total number of species. Ratings at the left (5, 3, 1) indicate whether species richness at a given site on a stream of a given order approximates, is somewhat less than, or is far less than the species richness expected for an "excellent" fish community in that region (from Fausch et al. 1984).

Karr (1981) tentatively assigned qualitative labels to total IBI scores, thus creating five classes of integrity ranging from excellent to very poor (Table 2). When repeated sampling at a site failed to produce fish, that site was assigned to a sixth class: no fish. Since Karr left undefined ranges of scores between classes, decisions are not made solely on the basis of total IBI scores. Instead, integrity classes are assigned in light of the attributes listed in Table 2 and with careful consideration of the expectation criteria.

Sampling Methods and Data Quality

Collection methods must be standardized to ensure the quality of data, and a sample must accurately reflect the fish community present in a stream reach at a specific time. Collecting several samples from a site on a given date is also a useful practice, but these samples should never be combined for IBI analysis.

Four problems in sampling stream fishes particularly affect the accuracy of the data, especially when historical data are used. First, the purpose for which the data were collected governs the nature of the data. Fish captured for taxonomic purposes, for example, are usually identified correctly but may not be accurately counted; species very common to a region may be ignored. Conversely, fish captured for purposes of fishery management will probably be counted, but small nongame species may be ignored or lumped into such categories as "forage fish." Second, sampling gear, water conditions, and fish behavior affect the accuracy of the sample. Certain species are difficult to capture with standard electrofishing and seining gear. Finding darters, for example, requires the thorough disturbance of riffles, and catfishes are often best sampled at night. High flows or turbid water, on the other hand, impair sampling efficiency for all species. Third, the range of habitats sampled has a major effect on data collection, and often the entire range of riffle, pool, and extra-channel habitats is not sampled, especially when large rivers are surveyed. Fourth, atypical samples result when unrepresentative habitats are adjacent to the sampling site. Species richness near bridges or near the mouths of tributaries entering large rivers, lakes, or reservoirs is, for example, more likely to be characteristic of larger-order habitats than the habitat under consideration (Fausch et al. 1984). Each of these four major sampling problems should be reviewed before data for IBI calculations are collected and especially when the use of historical data is being considered. Hendricks et al. (1980) have outlined a number of other problems in sampling stream fishes.

In spite of the care given to sampling design and sampling procedures, biologists must exercise judgment to ensure that a sample is representative. Gear, for example, must be capable of sampling all species in proportion to their relative abundance. Seines seem to be the best tool for sampling small, relatively simple streams. As streams increase in size and structural complexity, however, the efficiency of seines declines and more sophisticated equipment such as electric seines, which improve the sampling of undercut banks, log jams, and rock areas, may be required. Boat-mounted electrofishing equipment and, in some cases, hoop

or gill nets or rotenone are used. Such equipment, however, may decrease the number of smaller species captured. Indeed, most sampling procedures do not effectively capture fish less than 20 mm in length, a group that is also difficult to identify. We therefore usually recommend excluding fish in this size range, fish which generally prove to be young-of-the-year (Angermeier and Karr 1986). This policy lowers sampling costs and reduces the need for time-consuming laboratory studies.

Sampling effectiveness also varies according to the species of fish being sampled (Larimore 1961), their size (Jung and Libosvarsky 1965), visibility in the water and flow conditions of the stream (Paloumpis 1958), habitat structure (Gorman and Karr 1978), and a variety of other environmental factors (Cleary and Greenbank 1954; Mahon 1980). Nevertheless, a basic premise of IBI is that the entire fish fauna has been sampled in its true relative abundances without bias toward taxa or size of fish. As this assumption is relaxed, the reliability of inferences based on the IBI is reduced. Even with some reduced sampling rigor, however, IBI-based information can be used in making decisions about the management of water resources—a striking contrast to the "data-rich, information-poor" situation that often results from the routine monitoring of water quality. Too often, data from routine monitoring accumulates but analysis and the reporting of that analysis do not occur (Ward et al. 1986).

The size of the sample reach is another important consideration. In our experience, sample reaches of 100 m are sufficient in structurally simple headwater streams. In larger streams, selecting several contiguous riffle-pool sequences rather than relying on a standard length may be more appropriate. When electrofishing equipment is employed in larger rivers, samples should be taken in units of 0.5 to 1.0 km (Gammon et al. 1981). In brief, the length of the sample reach should be long enough to include all major habitat types—for example, riffles, pools, and backwater areas. Distances of 11 to 15 stream widths are generally adequate to sample two cycles of habitat (Leopold et al. 1964). In addition, the location of the site should be precisely recorded so sampling can be repeated in the future.

Selecting the appropriate time of year for sampling is critical. Our experience suggests that no single best period can be defined. In general, periods of low to moderate stream flow are preferred and the relatively variable flow conditions of early spring and late summer/autumn avoided. Species richness, for example, tends to be higher later in summer due to the presence of young-of-the-year of rare species, but this and other young-of-the-year problems can usually be avoided by sampling before late summer. Similarly, samples of limited area may be less variable in early summer than comparable samples taken later in the year.

Not only must these sampling guidelines be followed when data are collected specifically for use with IBI, but they must also be applied rigorously to historical data sets. Some data, therefore, may be rejected, especially when other data are available from the same area. The ultimate arbiter of the quality of a sample is a competent ichthyologist or aquatic ecologist who is familiar with the local fish fauna. In addition, users of IBI may want to confer with regional resource managers.

Validity of the Index of Biotic Integrity

Much has been written in recent decades about biomonitoring. Unfortunately, this literature has focused on the use of biomonitoring to detect biological effects of chemical pollutants (for example, Herricks and Cairns 1982; Herricks and Schaeffer 1985) or, in the case of macroinvertebrates, on oxygen depletion as a result of organic enrichment. As the earlier background discussion made clear, however, a number of factors other than toxic chemicals affect biotic integrity. Biomonitoring programs, therefore, must be designed to detect those forms of degradation, particularly because biotic integrity can sometimes be significantly improved without expensive chemical treatment. Herricks and Schaeffer (1985) defined six criteria that programs of biomonitoring should satisfy if they are to prove valid. Each of those criteria is summarized below along with a statement of the extent to which IBI satisfies it.

Criterion 1. The measure must be biological. IBI is based solely on the biological attributes of a water resource system and therefore meets this criterion.

Criterion 2. The measure must be interpretable at several trophic levels or provide a connection to other organisms not directly involved in the monitoring. The diversity of the three IBI metrics related to Trophic Composition, (metrics 7, 8, and 9, Table 1) assures that this criterion is met. Fish are affected by the availability of food (for example, benthic invertebrates) and by predation rates (presence of top carnivores), and IBI assesses both of these community attributes.

Criterion 3. The measure must be sensitive to the environmental conditions being monitored. IBI meets this criterion in broad-scale monitoring in ways not previously attained by indicators sensitive only to toxic chemicals or to oxygen depletion from organic enrichment. As we have shown in a number of published tests (Appendix III), IBI has a general sensitivity to many forms of degradation, including toxic chemicals and alterations of habitat and flow.

Criterion 4. The response range of the measure must be suitable for the intended application. IBI has demonstrated a sensitivity to small, even subtle, changes and to a broad range of conditions. This attribute derives from the diversity of its metrics and their varying ranges of sensitivity (Fig. 2).

Criterion 5. The measure must be reproducible and precise within defined and acceptable limits for data collected over space and time. When careful field methods are followed, IBI satisfies this criterion. Documentation for this statement is found, for example, in Figures 5 through 9.

Criterion 6. Variability of the measure must be low. Variability in IBI values for a given site may come from three sources: sampling imprecision or inadequacy (Criterion 5), natural variation over time due to climatic or other effects, and anthropogenic variation. As demonstrated here and in our other publications, IBI is sensitive to both natural and anthropogenic variation. An unfinished task, however, is to determine the magnitude of both sampling and natural vari-

ation. As Herricks and Schaeffer (1985) note, "The concern is not that a measure may be variable, but that the nature of that variability be well understood."

Application of IBI

Representative Examples

As noted earlier, IBI has been used widely by state and federal agencies throughout the United States and in Canada. In this section we present representative examples in graphic form. For more detailed information, the original sources should be consulted.

Five examples illustrate spatial variation within a watershed due to wastewater effluent: Figures 5, 10, 11, 13, and 14. Spatial variation due to variation in habitat quality is shown in Figures 5, 6, 7, 10, and 11. Temporal variation within a watershed is illustrated in Figures 7, 8, 9, and 13. Variation among watersheds is shown in Figure 10, the effects of an single pollutant in Figure 12, and the effects of three types of wastewater treatment in Figure 13. The remaining two figures show the family of maximum species richness lines that we have defined to date (Fig. 15) and the relationship between the quality of a water resource and its variability at several sites in two watersheds (Fig. 16).

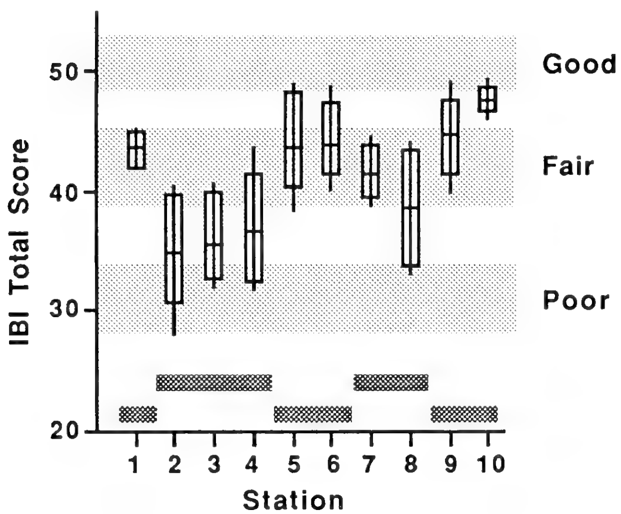

Fig. 5. IBI total scores at ten stations during 1978-80 on Big Ditch, a channelized, third-order stream in east-central Illinois with a moderate (1.5 m/km) gradient (from Karr et al. in press). Channel characteristics include homogeneous shallow raceways; sand, gravel, and rock substrates; and no riparian vegetation. Land use in the Big Ditch Watershed is 90% row crops. A municipal wastewater input just above Station 2 introduces 75 million liters per day of wastewater into Big Ditch.

IBI classified three sites below the wastewater input (stations 2, 3, and 4) and two downstream sites with poor instream habitat (stations 7 and 8) as poor to fair. Station 1 above the wastewater input and stations 5 and 6 and 9 and 10 were classified as fair to good. These stations are not impacted by the wastewater and have habitat of fair or good quality for areas in an agricultural watershed.

Shaded bars at the bottom of Figure 5 indicate groups of sites for which mean's are statistically indistinguishable (p <0.05, Student-Newman-Keuls Test). The vertical line that divides each station bar shows the range; the bar represents a standard deviation above and below the mean.

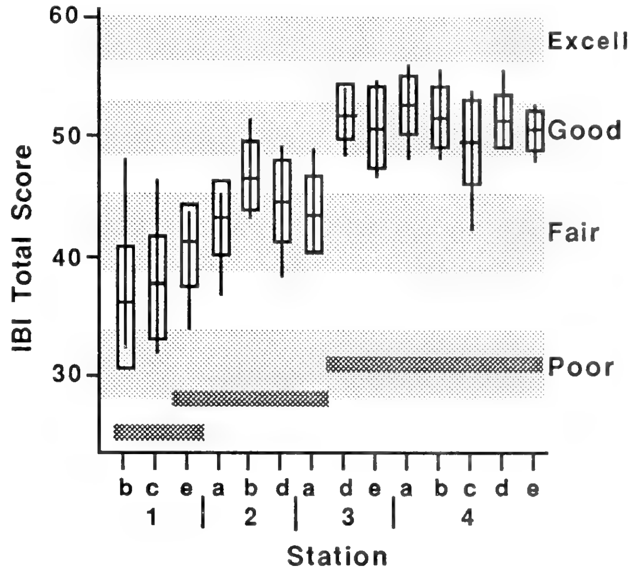

Fig. 6. IBI total scores for fourteen sites during 1978-80 in Jordan Creek, a first-to-third-order stream in east-central Illinois with four distinct stream segments (from Karr et al. in press). These four segments are characterized below.

Environmental Factor	Segments 1	2	3	4
Riparian habitat	none; cultivation to stream edge	8-10 m strip	pasture on riparian strip	10-400 m of forest
Gradient	0.65 m/km	0.72 m/km	0.95 m/km	4.0 m/km
Instream habitat	uniform raceway	poorly structured pools and raceways	well-developed pools	rocky riffles, well-developed pools
Channel history	recent channelization	old channelization	relatively natural	unchannelized
Substrate	unstable silt	silt and sand	sand and gravel	sand, gravel, rock
Watershed topography	level	moderately rolling	rolling	rolling

IBI identified three groups of sites in Jordan Creek. Upstream stations (1b and 1c) with the severest habitat modification had significantly lower IBI total scores than 2a to 3a. Highest IBI scores were at downstream stations 3d to 4e where the stream channel had not been reconstructed.

Shaded bars at the bottom of Figure 7 indicate sites for which means are statistically indistinguishable (p<0.05, Student-Newman-Keuls Test). The vertical line that divides each station bar shows the range; the bar represents a standard deviation above and below the mean.

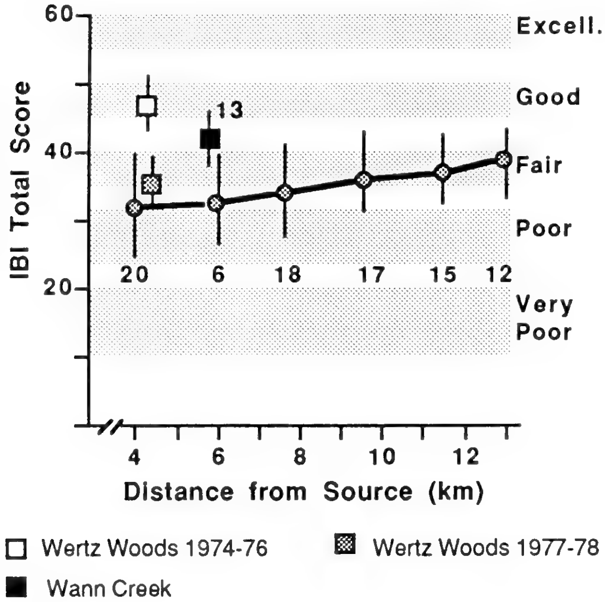

Fig. 7. IBI total scores (mean ± standard deviation) at six stations along main Black Creek Channel in Allen County, Indiana. Also shown is a site at Wertz Woods before (1974-76) and after (1977-78) upstream channel work in Wertz Drain. A Wann Creek site (Station 13) adjacent to the Black Creek Watershed was selected as a control site (1974-78) because it was not impacted by the Black Creek Sediment Control Project (from Karr et al. in press).

Upstream stations (20, 6, 18) are classed as poor to fair and downstream stations are classed as fair. Higher downstream values are due to proximity to the Maumee River and to more rolling topography that has protected near-channel habitats from the more pervasive agricultural impacts in the upstream area. The Wertz Woods reach is in a small woodlot protected from channelization. Following channelization upstream of Wertz Woods, IBI values declined sharply (see also Fig. 8). The Wann Creek site (Station 13) outside the Wertz Woods Watershed was comparable in size to Station 6 but usually had higher IBI scores due to lack of recent channelization.

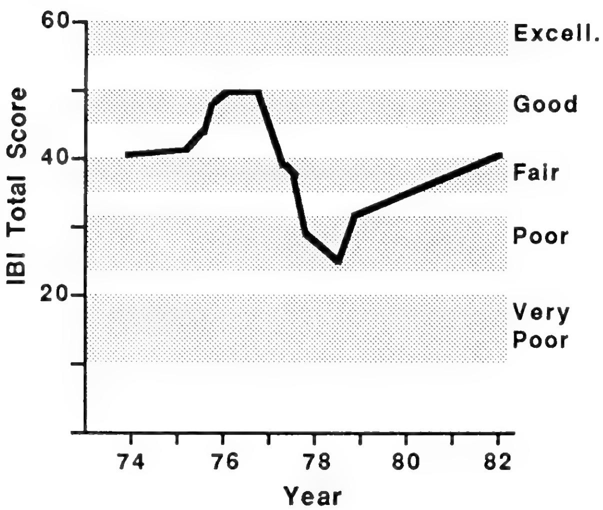

Fig. 8. Changes in IBI total scores over time in Wertz Drain at Wertz Woods, Allen County, Indiana (from Karr et al. in press). Wertz Woods, a small woodlot in the Black Creek Watershed, had relatively high IBI scores for a first-order stream in an area of intensive agriculture during 1974-76 as a result of good habitat quality (sinuous channel, well-developed pools and riffles, trees shading the channel). Although this site was not intentionally modified, a poorly executed bank stabilization effort upstream during 1976 resulted in the transport of sediment into this site. As a result of sedimentation in Wertz Woods, habitat quality deteriorated as did the resident fish community. IBI scores clearly indicate that decline and a slow improvement in following years.

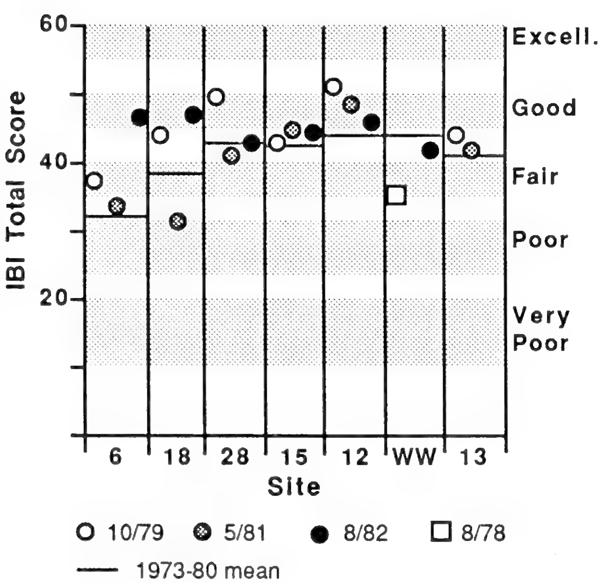

Fig. 9. Summary of IBI total scores at six sites in the Black Creek Watershed, Allen County, Indiana, and at Station 13 in Wann Creek. WW indicates Wertz Woods, a site discussed in greater detail in the legends to Figures 7 and 8. Despite massive expenditures to reduce the impact of agriculture in the Black Creek Watershed, most sites had IBI values at the end of the study, which ran from 1979 through 1982, that were near the long-term mean from 1973 to 1980 (from Karr et al. in press).

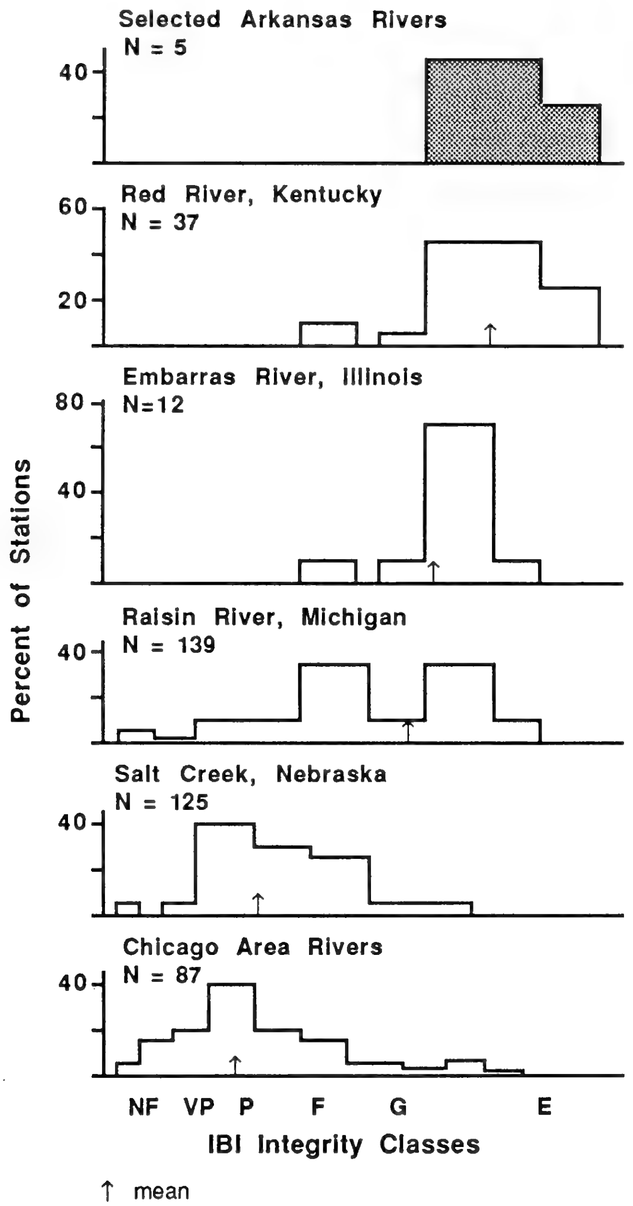

Fig. 10. Distribution of sites by IBI integrity classes in six midwestern regions or watersheds (modified from Fausch et al. 1984). Definitions of the six integrity classes are given in Table 2. IBI values in Chicago area streams are well below (90% fair or below) those of the less disturbed Red River Watershed, located partly in the Daniel Boone Wilderness, Kentucky, where 92% of the sites ranked "good" or above. The mean IBI value and range varies widely within watersheds (except in Arkansas where sample sites were selected to represent the best quality sites from each major region of the state) and between watersheds.

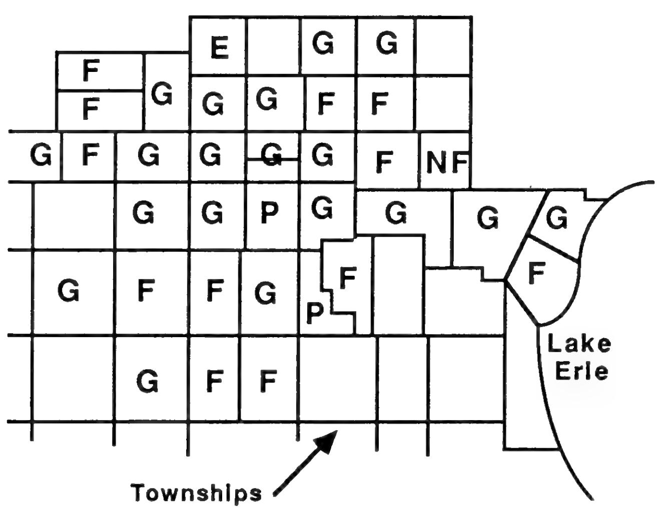

Fig. 11. IBI integrity classes for each township in the Raisin River, Michigan, Watershed. Areas with low IBI values are associated with larger towns, extensive agricultural areas, and feed lots. This type of geographic analysis can be used to define regions where additional study is needed to pinpoint degradation and to identify its causes and to suggest where regulatory activity should be increased. Definitions of integrity classes are given in Table 2.

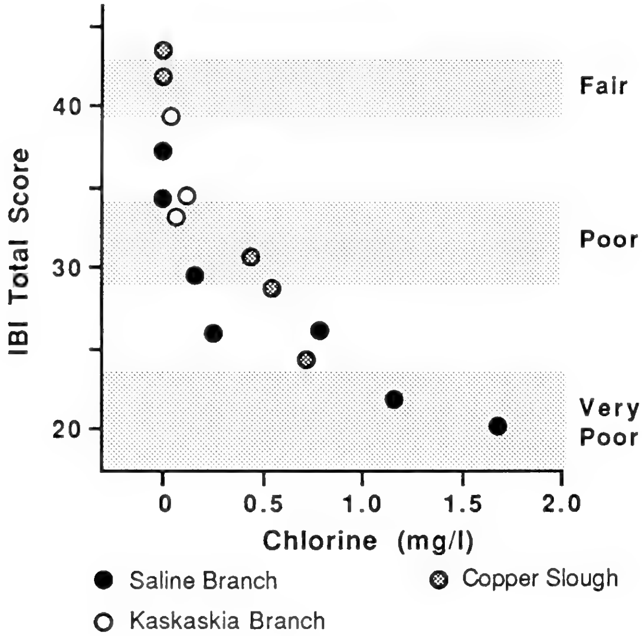

Fig. 12. IBI total scores versus chlorine content (mg/l) in three streams in east-central Illinois with wastewater inflow from secondary treatment with chlorination. Note the significant declines in IBI scores as residual chlorine concentration increases (from Karr et al. 1985a).

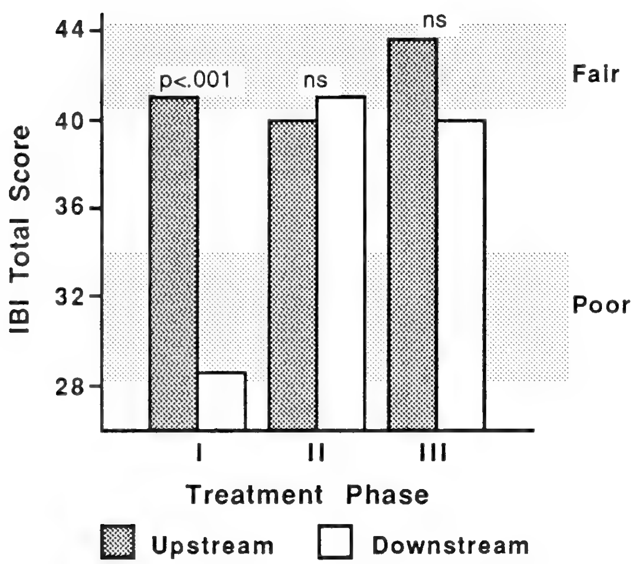

Fig. 13. IBI total scores for stations upstream and downstream of wastewater treatment effluent in Copper Slough, east-central Illinois. Phase I-Standard secondary treatment; Phase II-Secondary treatment without chlorination; Phase III-Secondary treatment without chlorination but with tertiary nutrification.

With chlorination (Phase I), IBI total scores are much lower downstream than upstream of effluent inflow. Upstream and downstream sites do not differ statistically following the removal of chlorine from secondary effluent (Phase II). The addition of expensive tertiary denitrification (Phase III) does not increase biotic integrity (from Karr et al. 1985a).

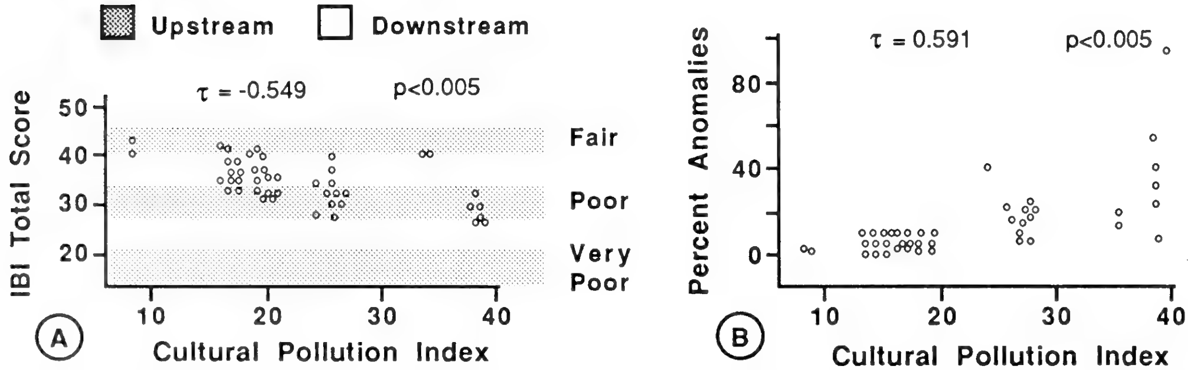

Fig. 14. Relationship between an index of cultural pollution for a series of streams in West Virginia and (A) IBI total scores and (B) the proportion of fishes with external anomalies (modified from Leonard 1984 and Leonard and Orth 1986). The cultural pollution index indicates levels of residential sewage loadings and mine drainage on streams. Other forms of degradation (e.g., habitat destruction and siltation) were relatively unimportant in these streams. IBI total scores declined significantly as cultural pollution increased. A clear relationship between cultural pollution and frequency of anomalies in fish was found.

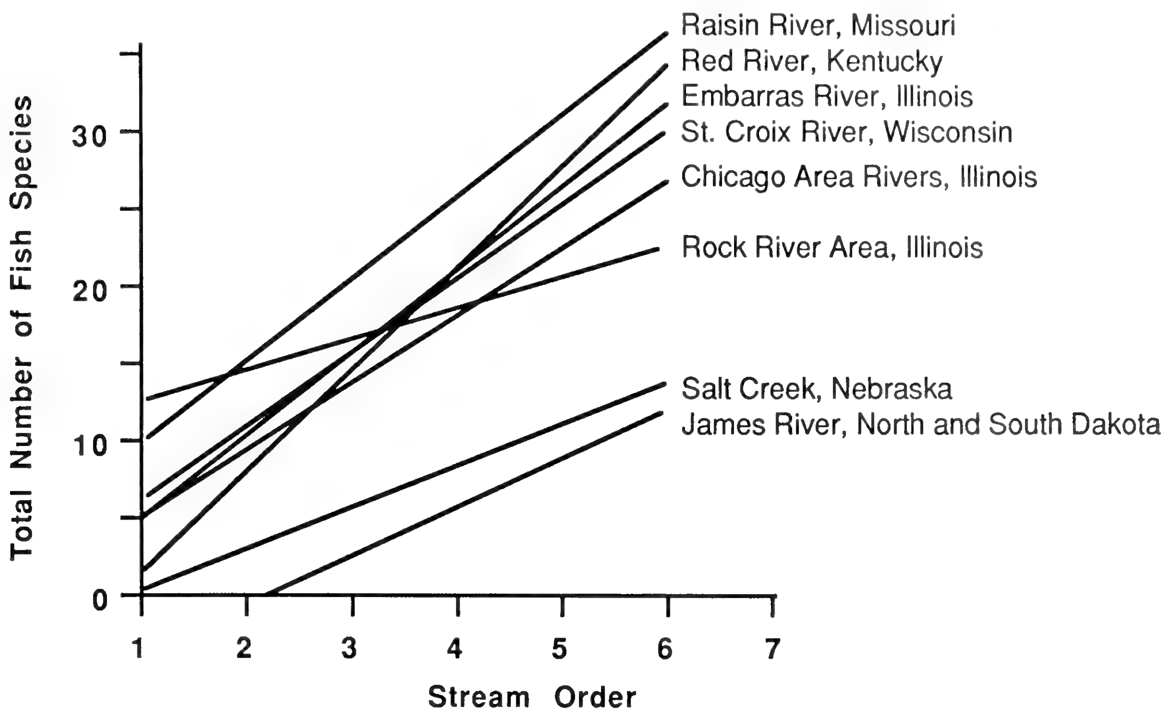

Fig. 15. Lines of maximum species richness for stream order using historical data from midwestern streams (modified from Fausch et al. 1984). These lines, which are used to determine the expected number of species at a site, also show that fish communities change predictably across regions and stream orders.

Although the lines for multiple species richness in these eight watersheds differed, they generally form two groups. The uppermost group is a set of woodland watersheds in the eastern Midwest. These six watersheds generally have more species at any stream order than do Great Plains streams, the lower group of two watersheds. Within each group, lines of maximum species richness tend to be similar. Further research will show whether this trend also holds with regard to watershed area. Application of these lines to regions other than those studied is inappropriate without intensive study of those regions.

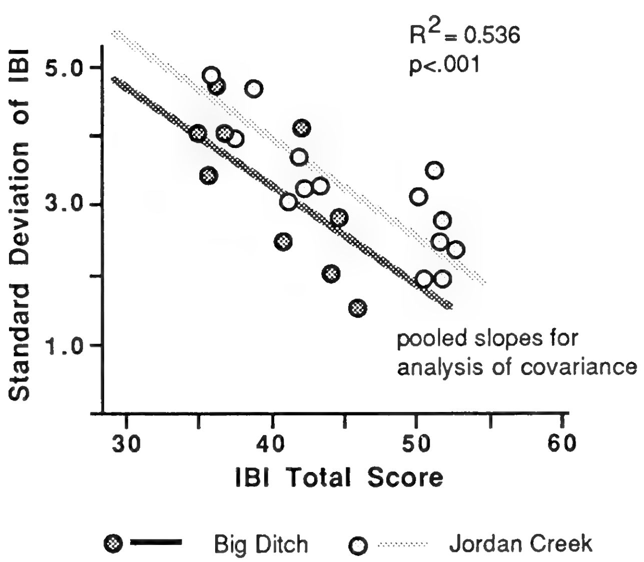

Fig. 16. Standard deviation of IBI as a function of IBI total scores in Jordan Creek and Big Ditch, east-central Illinois (from Karr et al. in press). Sites of high quality with high IBI scores in each stream were less variable over time than were sites of lower quality. Good sites, therefore, are more likely to be ranked near their mean than are poor sites. As a result, the overrating of poor sites is more likely than the underrating of good sites.

Inappropriate Uses

Any tool can be misused, and if the limitations of IBI are not recognized, it can be misapplied. The discussion that follows summarizes abuses to which IBI is subject.

General cautions. Management decisions based on IBI must be made with the guidance of a fish biologist familiar with IBI and with local fish fauna and watershed conditions. This point cannot be overemphasized. The use of IBI by individuals without biological training is as inappropriate as the use of econometric or engineering tools by those without specialized training.

Another potentially dangerous practice is turning IBI calculations over to a computer software package. A major advantage of IBI is its ability to integrate and summarize the collective wisdom of biologists. Computer programs, on the other hand, tend to overemphasize numerical data and minimize evaluation and interpretation, thus eliminating an important link in the chain of decision making. Because the expectations for a fish community vary with stream size (watershed area) and regional zoogeography, a considerable investment of time is required to define expectation criteria and to collect, collate, and interpret data from sampling sites. An invalid IBI profile can be calculated with or without a computer in only a few minutes. An accurate profile can be drawn only after the careful interpretation of all relevant information.

Finally, management at the watershed level is essential if the problems indicated by low IBI scores are to be solved (Karr and Schlosser 1978). Management practices that merely improve metric scores temporarily do not improve biotic integrity. The stocking of piscivores (top carnivores), for example, may increase a local IBI value temporarily but if these fishes have little chance of long-term survival, the measure is pointless.

Sampling cautions. Representative samples are essential. Among the most common problems associated with sampling are reliance on stream reaches that are too short and gear that is ineffectual for certain species or habitats. How to avoid these hazards has been discussed earlier. Principles of channel hydrology suggest that reach length should be at least 11 to 15 channel widths or a minimum reach length of 100 m in small streams. A good rule of thumb is to sample two cycles of representative riffle-run-pool habitats and other associated channel habitats. Since some fishes avoid capture when standard gear is used—especially riffle species, nocturnal species, and some top predators—gear must be used that minimizes sampling bias. All species present in the sample reach must be captured if IBI is to function reliably. Further, the proportional representation (abundance) of a species in the sample must be similar to its presence in the stream. As sampling bias increases, inferences based on IBI data become less reliable. When sampling bias exists, it should be acknowledged and the interpretation of data made in the light of that bias.

When a representative sample has been taken, every effort must be made to identify and count all individuals in the sample. Biologists charged with the management of sport or commercial fisheries often cannot or do not identify or count most nongame fish. Similarly, icthyologists often do not count fish of all species present during faunal surveys and tend to sample the most diverse faunas available. As a result, data from such sources may be badly skewed. Similarly, historical data sets, although they provide considerable insight about present and past faunas, must be used cautiously. Like modern samples, historical samples may not be representative because of the gear used, because of identification and enumeration inadequacies, or because of poorly chosen sampling locations. Samples taken at sites near bridges, dams, or other habitat anomalies, for example, should be avoided because they are often unrepresentative of regional conditions.

Finally, the lines of maximum species richness used in IBI calculations are based on species collected at one site on one day—one-sample richness. Collections taken at one site on different dates, therefore, cannot be combined and used. The accuracy of metrics 1-6 depend on one-sample richness.

Cautions regarding the interpretation of IBI data. The importance of professional judgment during the collection of samples, the development of expectation criteria, the assignment of metric scores, and the interpretation of those scores cannot be overemphasized. On occasion, for example when historical data are used, information for certain metrics may be unavailable. In our experience, missing or inadequate data are most likely to occur for metrics 11 and 12, hybrids and disease. Omitting a metric would alter the overall range of IBI and thus would require a rescaling of IBI values. At present, we believe that the best approach may be to assign a rating of 5 to metrics with missing data, a conservative alternative since it inflates site quality in degraded locations.

IBI, of course, is not the last word in the management of water resources. Instead, it is a tool that aids in the interpretation of complex biological data and a method that integrates physical and chemical data. In practice, we believe that specific IBI scores should be minimized and the integrity classes (excellent to very poor) emphasized. IBI scores for given sites are always relative to one another and have no absolute meaning.

Our analyses of temporal and spatial variability in IBI sampling and subsampling suggest that total IBI scores should differ by at least 4 points before a change in site quality can be said to exist. This range varies, however, and differences greater than 4 may be required when streams are degraded, when sampling controls were poor and samples are unreliable, and when larger streams are being sampled.

Finally, for a variety of reasons, caution must be exercised when comparing streams from different geographic regions. Qualitative labels (excellent to very poor) may be used in making comparative statements but quantitative IBI scores cannot. IBI cannot be used in cookbook fashion as indexes of species diversity are sometimes used. When used correctly, however, it provides a synthesis of biological information not possible with other indexes.

Future Uses

The widespread adoption of IBI by state and federal agencies as a tool for water resources managers suggests a useful future for IBI. Its continued use in the making of informed decisions regarding the nation's water resources, however, depends on four major developments: the initiation of training courses, the creation of substitute metrics to accommodate regional differences, the continued study of natural and anthropogenic variations in biotic integrity so that IBI data can be interpreted with even greater precision, and the documentation of the distributional (statistical) properties of IBI.

Two levels of training are required. First, biologists already familiar with the ecological concepts on which IBI is based must gain an understanding of how and why those concepts are used in IBI. Second, water resource managers at all regulatory levels and from all disciplines will need to understand the importance of the direct assessment of biotic integrity. Their attention has long been focused on assessments of water quality; IBI focuses their attention on the quality of water resources.

IBI is potentially useful in a wide range of aquatic and marine environments across geographic areas and with taxa other than fish. Equivalent metrics, however, must be developed. Lake communities, for example, are organized in ways that differ from those of small streams. Because IBI was developed for use in small to medium-sized midwestern streams, its metrics cannot be applied to the study of lake communities. The principles on which those metrics were developed, however, can be used to create equivalent ecological metrics for lake communities. Similarly, IBI can be adapted to other taxa—benthic invertebrates, for example—and to regions with other biogeographic and evolutionary histories—for example, California or New England—when the ecological structure of those communities has been carefully studied. Efforts are already underway to adapt IBI to larger rivers, to estuarine and lake environments, to streams outside the Midwest, and to other taxa.

The range of natural variation in stream communities must be defined so that techniques to distinguish natural from anthropogenic variation can be developed. As noted earlier, variation in an index value does not make the index useless. Rather, we must develop a clear understanding of and appreciation for that variation. Research to accomplish this goal is roughly the analog of defining criteria and standards for specific pollutants. This task is especially important because considerable time and money can be saved by not "fixing" natural variation.

A long-term goal in the use of IBI and related biological monitoring tools should be the treatment of the index as a statistic that has sampling and other sources of variability. The distributional properties of IBI must be documented.

In addition to these four major developments, three more minor aspects of IBI are important in its future use: the treatment of exotic species, the scoring of metrics related to trophic composition, and the handling of one-species guilds.

Fish communities of the Midwest have been invaded by a few exotic species, carp, for example, but these are a small proportion of the species in the rich fish fauna of midwestern rivers. A major proportion of the fish assemblage in western streams, however, may be exotics. At least one study indicates that undisturbed streams contain no exotic species but that an abundance of exotic species characterizes the most severely disturbed stream reaches (Leidy and Fiedler 1985). Exotics (the number of species, the percent of species, or the percent of individuals) might be used to develop a valuable metric for use in areas with relatively depauperate communities.

The general relationship between productivity and trophic composition of stream communities is firmly established (Vannote et al. 1980), and this relationship is incorporated into several IBI metrics. To date we have used the same expectation criteria for Trophic Composition (metrics 7, 8, and 9) in all regions. Further research, however, is needed to determine if the expectation criteria for these three metrics should vary, for example, with stream size, with region, and between cold- and warm-water streams.

Such major taxa as sunfishes and darters were selected because they were represented by a number of species in most midwestern watersheds. In some regions, however, no multispecies taxa may be present and substituting abundance information for metrics 2 and 3 may not be appropriate. In western streams with one very abundant scuplin, for example, metric 2 (number and identity of darter species) might be replaced by a metric giving the percent of individuals as sculpins.

Acknowledgments

The development of IBI was stimulated by the participation of James R. Karr in a sediment control project initiated almost fifteen years ago in Indiana's Black Creek Watershed. Since then, numerous colleagues have commented on the relationship we saw between the organization of the biological communities of running water ecosystems and the goals of clean water legislation. IBI could not have been developed without those comments and the insights developed in recent years by fishery biologists, aquatic ecologists, and water resources planners. Their contributions before the development of IBI and their comments since have contributed immeasureably to this document. Careful editorial work by Audrey Hodgins of the Illinois Natural History Survey clarified many inconsistencies in the original manuscript.

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Appendix I. Calculation of IBI Scores with Example Data Sets.

Eight steps—beginning with the development of expectation or scoring criteria for a given site and ending with the assignment of an integrity class to that site—make up the logical sequence of IBI calculations. Data from three sample reaches in the Embarras River Basin in Illinois (Fausch et al. 1984) are used to illustrate these steps. We assume that these data accurately reflect the species present in the fish community and their relative abundances.

1. Develop expectation criteria for the watershed under study. Later, these criteria will be used to rate the data accumulated for each IBI metric.

Species richness and composition (metrics 1-6). Scoring criteria for the five metrics that indicate species richness (total number of species and number and identity of darter, sunfish, sucker, and intolerant species as listed in Appendix Table 1) are the most difficult to establish. Following the methods used by Fausch and his colleagues (1984) to determine maximum species richness, plot the number of species as a function of stream size (either in terms of stream order or watershed area) at selected sites within a watershed. These points, as shown in the text (Fig. 4), produce a right triangle the hypotenuse of which, when fit by eye, forms the upper bound of collections taken at 95% of the sites. This line of maximum species richness is assumed to define an "excellent" fish community and is assigned a value of 5 for purposes of rating metrics 1-5. Divide the area beneath this line into thirds and assign those lines values of 3 and 1 in descending order of richness. These values, in turn, are used to rate metrics for sites of lower species richness.

Green sunfish was selected as a midwestern species that is common in relatively undisturbed areas but becomes very abundant in degraded situations. The proportions of individuals as green sunfish assigned to metric ratings of 5, 3, and 1 are given in Appendix Table 1.

Trophic composition (metrics 7-9). Because the food base is central to the maintenance of a community, information about trophic composition is important in IBI calculations. We have found no evidence that threshold values for metrics 7-9 vary regionally or as a function of stream size up to the sixth order, with the possible exception of first-order streams. We therefore recommend the threshold values shown in Appendix Table 1 unless future studies challenge their generality. In regions where insectivorous cyprinids do not dominate (as they do in the Midwest), we suggest that proportion of all insectivores be used instead with the following scoring criteria: 1 = 0-40%, 3 = >40-80%, 5 = >80-100%.

Fish abundance and condition (metrics 10-12). When historical data are used, we suggest that calculations for metric 10, number of individuals in the sample, be based on catch per unit of effort, with effort expressed as surface area, length of stream reach, or time spent sampling. Sites with high capture rates are likely to be of higher quality than sites with low capture rates when other conditions at the two sites are similar. When data are obtained from current samples, a density estimate (number of individuals per 100 m^2) should be used for this metric. Miller et al. (unpublished manuscript) have developed an alternative based on the decrease in density as watershed area or stream order increases. They suggest a regression of density on watershed area and follow the method (Fausch et al. 1984) that we used to draw lines of species richness. They calculate the

Appendix Table 1. Scoring criteria used to rate three third-order stream sites on the Embarras River, Illinois.

		Scoring criteria		
Category	Metric	5	3	1
Species richness and composition	1. Total number of fish species	≥10	9-4	3-0
	2. Number and identity of darter species	≥3	2-1	0
	3. Number and identity of sunfish species	≥2	1	0
	4. Number and identity of sucker species	≥2	1	0
	5. Number and identity of intolerant species	≥3	2-1	0
	6. Proportion of individuals as green sunfish	<5%	5-20%	>20%
Trophic composition	7. Proportion of individuals as omnivores	<20%	20-45%	>45%
	8. Proportion of individuals as insectivorous cyprinids	>45%	45-20%	<20%
	9. Proportion of individuals as piscivores (top carnivores)	>5%	5-1%	<1%
Fish abundance and condition	10. Number of individuals in sample	0-50	>50-200	>200
	11. Proportion of individuals as hybrids	0	>0-1%	>1%
	12. Proportion of individuals with disease or other anomaly	0-2%	>2-5%	>5%

upper 90% prediction limit and then trisect the area under this line to provide values for ratings of 5, 3, and 1.

Metrics 11 and 12, hybrids and disease, are most useful in assessing moderately to severely degraded sites. As is the case for metrics 7-9, we have detected no variation in expectations based on stream size or geographic region for metrics 11 and 12. We therefore recommend the scoring criteria shown in Appendix Table 1. When no data are available for metrics 11 and 12, we suggest arbitrarily assigning scores of 5.

2. Tabulate the number of fish of each species in the collection. List all species in taxonomic order following *A List of Common and Scientific Names of Fishes from the United States and Canada* (Robins et al. 1980). Then tally the number of individuals for each species for each site (Appendix Table 2).

3. Assign each species to a trophic guild according to its food habits. We used the following definitions to categorize the species listed in Appendix Table 2.

Insectivores (I): adult diet consists of more than 75% insects

Piscivores (P): adult diet consists of more than 75% fish

Herbivores (H): adult diet consists of more than 75% plant material obtained by grazing off substrates or feeding on vascular macrophytes

Omnivores (O): adult diet consists of more than 25% plant material and more than 25% animal material

Planktivores (Pl): adult diet consists of more than 75% zooplankton and/or phytoplankton

Appendix II lists the trophic guilds for many species that we have collected in midwestern streams. Lack of feeding data sometimes requires researchers to infer the food habits of certain species from morphological data. Regional references, primary literature, and biologists familiar with the area can also help determine the local food habits of a given species.

4. Identify intolerant species. Because of their inability to survive disturbance, intolerant species are among the first

Appendix Table 2. Fish species and their abundances at three sample sites on third-order stream reaches on the Embarras River, Illinois, including trophic guild and the identification of intolerant species. Trophic guilds and their abbreviations are defined in the text; IS indicates intolerant species.

Family Species	Trophic guild	Intolerant species	Number of individuals Site X	Site Y	Site Z
Cyprinidae					
Central stoneroller *(Campostoma anomalum)*	H	—	—	—	7
Silverjaw minnow *(Ericymba buccata)*	I	—	—	—	9
Central silvery minnow *(Hybognathus nuchalis)*	H	—	—	13	—
Golden shiner *(Notemigonus crysoleucas)*	O	—	6	—	1
Emerald shiner *(Notropis atherinoides)*	I	—	—	8	—
Striped shiner *(Notropis chrysocephalus)*	I	—	—	—	14
Ribbon shiner *(Notropis fumeus)*	I	—	—	—	1
Sand shiner *(Notropis stramineus)*	I	—	—	—	2
Redfin shiner *(Notropis umbratilis)*	I	—	—	—	183
Bluntnose minnow *(Pimephales notatus)*	O	—	1	11	43
Creek chub *(Semotilus atromaculatus)*	I	—	—	1	19
Catostomidae					
Creek chubsucker *(Erimyzon oblongus)*	I	—	—	—	3
Spotted sucker *(Minytrema melanops)*	I	IS	—	—	2
Ictaluridae					
Black bullhead *(Ictalurus melas)*	I	—	—	1	—
Cyprinodontidae					
Blackstripe topminnow *(Fundulus notatus)*	I	—	1	16	—
Poeciliidae					
Mosquitofish *(Gambusia affinis)*	I	—	1	—	—
Centrarchidae					
Green sunfish *(Lepomis cyanellus)*	I	—	2	—	—
Bluegill *(Lepomis macrochirus)*	I	—	1	4	—
Longear sunfish *(Lepomis megalotis)*	I	IS	—	—	4
Smallmouth bass *(Micropterus dolomieui)*	P	—	2	5	—
Percidae					
Greenside darter *(Etheostoma blennioides)*	I	IS	—	—	2
Rainbow darter *(Etheostoma caeruleum)*	I	IS	—	—	18
Johnny darter *(Etheostoma nigrum)*	I	—	—	2	1
Blackside darter *(Percina maculata)*	I	—	—	—	2

to be decimated. For the purpose of IBI calculations, we recommend that the intolerant class be restricted to the 5 to 10% of the species that are most susceptible to degradation of any type. As is the case in determining trophic classes, intolerant species can be determined from regional ichthyological references and from those with special knowledge of local fish fauna. Generally, intolerant species disappear before a site degrades to an IBI integrity class of "fair." The intolerant species found in the three sample sites on the Embarras River are shown in Appendix Table 2. Tolerance classes, however, may vary for species that have wide geographic ranges, and rare species, as was noted in the text, are not necessarily intolerant species.

5. Summarize the biological information available for each IBI metric. To simplify the scoring task, prepare a summary worksheet similar to the one shown in Appendix Table 3. Determine the number of taxa, the proportion of green sunfish, and the proportion of omnivores, insectivorous cyprinids, and piscivores. The total number of individuals in the sample should be included as should the proportion of hybrids and diseased individuals.

6. Rate each metric according to the corresponding scoring criterion developed for the site. Using Appendix Table 1, assign a score of 5, 3, or 1 according to whether the information accumulated for a given metric approximates, deviates somewhat from, or deviates strongly from conditions expected in an undisturbed stream of similar size in a similar geographical location.

7. Calculate the total score by adding the ratings assigned to the twelve metrics. The maximum total (60) indicates a site without perturbation. The minimum score of 12 is possible when all metrics reflect extreme degradation.

8. Convert the total score to one of five biotic integrity classes using the numerical ranges given earlier in the text (Table 2). If degradation is so severe that no fish are present, a sixth class—no fish—may be added.

Integrity classes provide useful labels that are easily understood by nonprofessionals; however, since the numerical ranges for these classes do not overlap, absolute total scores and the original metric data can be used by professionals to make additional inferences about the relative quality of a site. Attributes of the site can also be compared to the descriptive attributes used to characterize each integrity class (Text Table 2). As suggested by Karr (1981), a careful look at the species present in a collection provides additional insight. Large families contain species that can be ranked according to tolerance: green sunfish, white sucker, and johnny darter, for example, are more tolerant than longear sunfish, northern hog sucker, and banded darter. Sites dominated by the less tolerant species in these and other groups can be assessed accordingly.

Appendix Table 3. Summary worksheet for IBI calculations at three third-order stream sites on the Embarras River, Illinois.

Classification of data	Site X		Site Y		Site Z	
Number of species (metrics 1-5)						
Total	7	(3)[a]	9	(3)	16	(5)
Darters	0	(1)	1	(3)	4	(5)
Sunfishes	2	(5)	1	(3)	1	(3)
Suckers	0	(1)	0	(1)	2	(5)
Intolerants	0	(1)	0	(1)	4	(5)
Proportion of individuals (metrics 6-9, 11-12)						
Green sunfish	14%	(3)	0%	(5)	0%	(5)
Omnivores	50%	(1)	18%	(5)	14%	(5)
Insectivorous cyprinids	0%	(1)	15%	(1)	73%	(5)
Piscivores (top carnivores)	14%	(5)	8%	(5)	0%	(1)
Hybrids	—	(5)[b]	—	(5)	—	(5)
Diseased	—	(5)[b]	—	(5)	—	(5)
Total number of individuals in the sample (metric 10)	14	(1)	61	(3)	311	(5)
IBI total score	32		40		54	
Integrity class	poor		fair		good to excellent	

[a]IBI metric ratings (5, 3, 1) are given in parentheses.

[b]Since information on the incidence of hybrids and disease was not available, scores of 5 were arbitrarily assigned, as noted in step one (Appendix I) and in the text.

Appendix II. Trophic Guilds for Common Freshwater Fishes of North-central North America, Including the Identification of Intolerant Species. (Trophic guilds and their abbreviations are defined in the text; IS indicates intolerant species.)

Family Species	Trophic guild	Intolerant species
Clupeidae		
Gizzard shad *(Dorosoma cepedianum)*	O	—
Umbridae		
Central mudminnow *(Umbra limi)*	O	—
Esocidae		
Grass pickerel *(Esox americanus)*	P	—
Northern pike *(Esox lucius)*	P	—
Cyprinidae		
Central stoneroller *(Campostoma anomalum)*	H	—
Goldfish *(Carassius auratus)*	O	—
Common carp *(Cyprinus carpio)*	O	—
Silverjaw minnow *(Ericymba buccata)*	I	—
Brassy minnow *(Hybognathus hankinsoni)*	O	—
Central silvery minnow *(Hybognathus nuchalis)*	H	IS
Silver chub *(Hybopsis storeriana)*	I	—
Hornyhead chub *(Nocomis biguttatus)*	I	—
Golden shiner *(Notemigonus crysoleucas)*	O	—
Emerald shiner *(Notropis atherinoides)*	I	—
River shiner *(Notropis blennius)*	I	—
Striped shiner *(Notropis chrysocephalus)*	I	—
Common shiner *(Notropis cornutus)*	I	—
Blacknose shiner *(Notropis heterolepis)*	I	IS
Spottail shiner *(Notropis hudsonius)*	I	IS
Red shiner *(Notropis lutrensis)*	O	—
Rosyface shiner *(Notropis rubellus)*	I	IS
Spotfin Shiner *(Notropis spilopterus)*	I	—
Sand shiner *(Notropis stramineus)*	I	—
Redfin shiner *(Notropis umbratilis)*	I	—
Steelcolor shiner *(Notropis whipplei)*	I	IS
Suckermouth minnow *(Phenacobius mirabilis)*	I	—
Southern redbelly dace *(Phoxinus erythrogaster)*	H	IS
Bluntnose minnow *(Pimephales notatus)*	O	—
Fathead minnow *(Pimephales promelas)*	O	—
Blacknose dace *(Rhinichthys atratulus)*	I	—
Creek chub *(Semotilus atromaculatus)*	I	—
Catostomidae		
River carpsucker *(Carpiodes carpio)*	O	—
Quillback *(Carpiodes cyprinus)*	O	—
Highfin carpsucker *(Carpiodes velifer)*	O	IS
White sucker *(Catostomus commersoni)*	I	—
Blue sucker *(Cycleptus elongatus)*	O	—
Creek chubsucker *(Erimyzon oblongus)*	I	—
Northern hog sucker *(Hypentelium nigricans)*	I	IS
Smallmouth buffalo *(Ictiobus bubalus)*	I	—
Bigmouth buffalo *(Ictiobus cyprinellus)*	I/P	—
Black buffalo *(Ictiobus niger)*	I	—
Spotted sucker *(Minytrema melanops)*	I	IS
Silver redhorse *(Moxostoma anisurum)*	I	—
River redhorse *(Moxostoma carinatum)*	I	IS
Black redhorse *(Moxostoma duquesnei)*	I	IS
Golden redhorse *(Moxostoma erythrurum)*	I	—
Shorthead redhorse *(Moxostoma macrolepidotum)*	I	—

(Appendix II continued.)

Family Species	Trophic guild	Intolerant species
Greater redhorse *(Moxostoma valenciennesi)*	I	—
Ictaluridae		
Blue catfish *(Ictalurus furcatus)*	I/P	—
Black bullhead *(Ictalurus melas)*	I	—
Yellow bullhead *(Ictalurus natalis)*	I	—
Brown bullhead *(Ictalurus nebulosus)*	I	—
Channel catfish *(Ictalurus punctatus)*	I/P	—
Slender madtom *(Noturus exilis)*	I	IS
Stonecat *(Noturus flavus)*	I	—
Tadpole madtom *(Noturus gyrinus)*	I	I/S
Freckled madtom *(Noturus nocturnus)*	I	—
Flathead catfish *(Pylodictis olivaris)*	P	—
Aphredoderidae		
Pirate perch *(Aphredoderus sayanus)*	I	—
Percopsidae		
Trout-perch *(Percopsis omiscomaycus)*	I	—
Cyprinodontidae		
Blackstripe topminnow *(Fundulus notatus)*	I	—
Poeciliidae		
Mosquitofish *(Gambusia affinis)*	I	—
Percichthyidae		
White bass *(Morone chrysops)*	I/P	—
Yellow bass *(Morone mississippiensis)*	I/P	—
Centrarchidae		
Rock bass *(Ambloplites rupestris)*	I/P	IS
Green sunfish *(Lepomis cyanellus)*	I/P	—
Pumpkinseed *(Lepomis gibbosus)*	I	—

Family Species	Trophic guild	Intolerant species
Orangespotted sunfish *(Lepomis humilis)*	I	—
Bluegill *(Lepomis macrochirus)*	I	—
Longear sunfish *(Lepomis megalotis)*	I	IS
Redear sunfish *(Lepomis microlophus)*	I	—
Smallmouth bass *(Micropterus dolomieui)*	I/P	—
Largemouth bass *(Micropterus salmoides)*	I/P	—
White crappie *(Pomoxis annularis)*	I/P	—
Black crappie *(Pomoxis nigromaculatus)*	I/P	—
Percidae		
Western sand darter *(Ammocrypta clara)*	I	IS
Rainbow darter *(Etheostoma caeruleum)*	I	—
Fantail darter *(Etheostoma flabellare)*	I	—
Slough darter *(Etheostoma gracile)*	I	—
Least darter *(Etheostoma microperca)*	I	—
Johnny darter *(Etheostoma nigrum)*	I	—
Orangethroat darter *(Etheostoma spectabile)*	I	—
Banded darter *(Etheostoma zonale)*	I	IS
Yellow perch *(Perca flavescens)*	I/P	—
Logperch *(Percina caprodes)*	I	—
Blackside darter *(Percina maculata)*	I	—
Slenderhead darter *(Percina phoxocephala)*	I	IS
Sauger *(Stizostedion canadense)*	P	—
Walleye *(Stizostedion vitreum)*	P	—
Cottidae		
Mottled sculpin *(Cottus bairdi)*	I	IS

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High quality manuscripts dealing with any aspect of natural history will be considered for publication in one of the Illinois Natural History Survey series: *Bulletin, Biological Notes, Circular,* and *Special Publication.* Authors who are not employees of the Survey are required to pay printing costs. Manuscripts should follow the recommendations of the third edition of the *Council of Biology Editors Style Manual* except that journal names in the literature cited section are to be spelled in full. The Survey expects to publish only one or two manuscripts by non-Survey authors yearly.

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